Passage of Juvenile and Adult Salmonids at Columbia and Snake River Dams

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INTRODUCTION

Development of the Federal Columbia River Power System (FCRPS) on mainstem Snake and Columbia Rivers began with the completion of Bonneville Dam in 1938, and ended with the construction of Lower Granite Dam in 1975 (Fig. 1). Throughout the past six decades, many structural configurations and operational strategies have been tested to improve the survival of juvenile and adult salmonids (*Oncorhynchus* spp.) passing through the FCRPS. This report summarizes the information pertinent to the FCRPS as it is currently configured for each route of passage and life history, and discusses uncertainties associated with the existing database. The reader is referred to Mighetto and Ebel (1994) and Whitney et al. (1997) for historical reviews of studies conducted over the past decades to evaluate the causes of fish loss, various improvements tested, and programs implemented to improve fish condition and survival through the FCRPS.

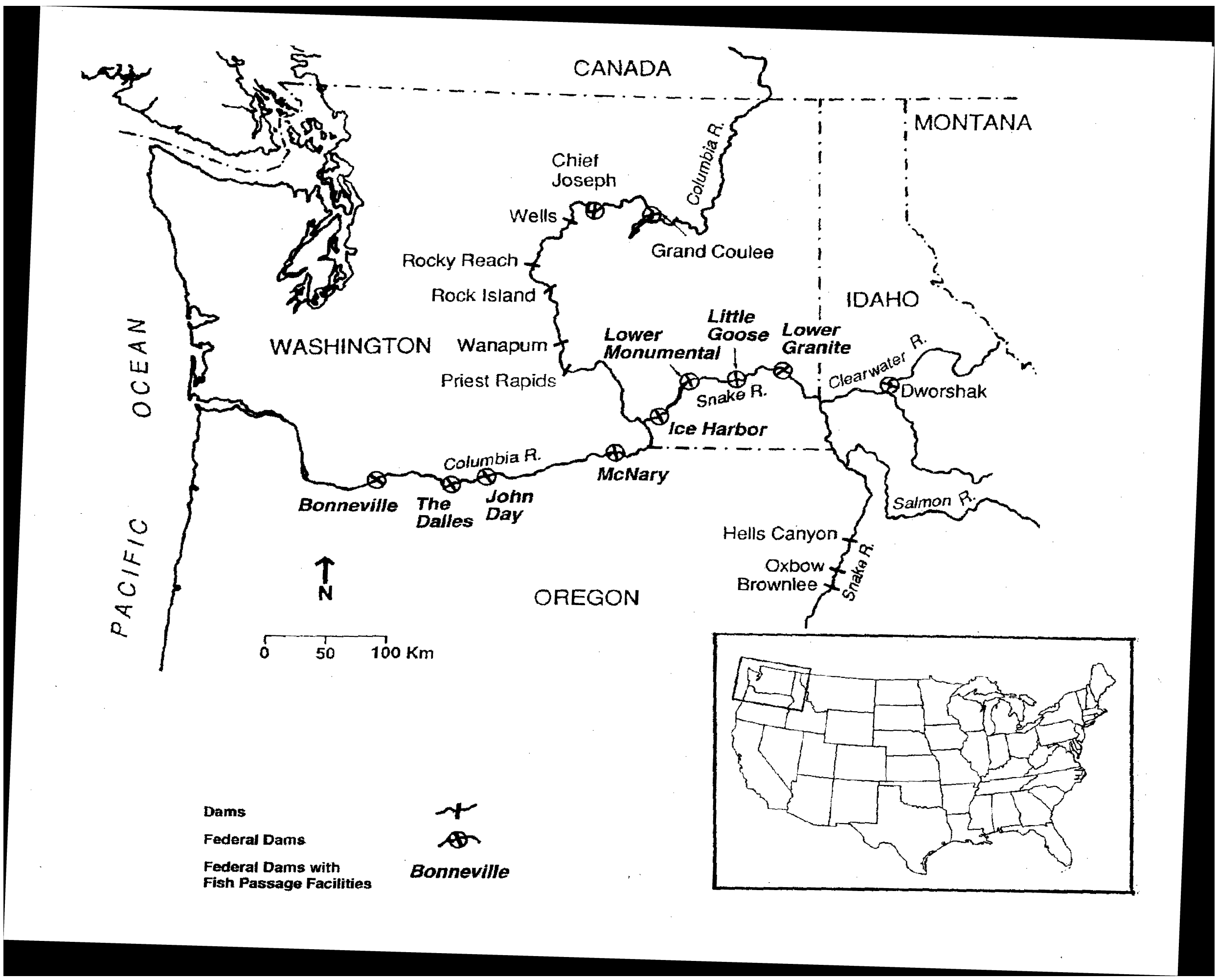

Figure 1. Federal Columbia River Power System.

JUVENILE PASSAGE THROUGH SPILLWAYS

Spill Management

Background

Spill has long been considered one of the safest routes of passage for juvenile salmonids at Snake and Columbia River hydroelectric projects. In the l940s, the U.S. Fish and Wildlife Service conducted survival studies at the newly constructed Bonneville Dam. After 7 years of releases of juvenile chinook salmon and the subsequent recovery of adults, the agency reported that spillway survival was 97% using pooled data and 96% using weighted averages, and that these values were about 10% higher than turbine passage survival (Holmes 1952). Studies conducted since indicate a similar trend, where survival through spillways was generally better than through turbines (Whitney et al. 1997). Based on this information, regional fishery managers have long regarded spill as the safest passage route for juvenile salmonids.

Despite the fishery agency positions in favor of spill for fish passage, the majority of spill at hydroelectric projects prior to the 1980s was involuntarily caused by river flows that exceeded the hydraulic capacity of the powerhouses. Voluntary spill reduced power production and was provided only for short periods to address specific passage situations. Much of the conflict over using spill for fish passage was based primarily on the fact that dams were authorized and operated for power production, not fish passage.

On 5 December 1980, the Pacific Northwest Electric Power Planning and Conservation Act created the Northwest Power Planning Council (Council) and required the establishment of a fish and wildlife program consisting of measures “to protect, mitigate, and enhance fish and wildlife affected by the development, operation, and management” of the hydroelectric facilities located on the Columbia River and its tributaries. In 1982, the Council released the first version of this program now known as the Columbia Basin Fish and Wildlife Program (Program).

This program and its amended versions contained measures that instructed the U.S. Army Corps of Engineers (COE) to “develop and implement a plan for spills which will achieve a level of smolt survival comparable to or better than that achievable by the best available bypass and screening systems” at several mainstem dams including John Day, Ice Harbor and Lower Monumental Dams (NPPC 1982).

In the 1984 Program amendment, a similar requirement for interim spill was added for The Dalles and Bonneville Dams (NPPC 1984). The 1984 amendment also established a 90% survival goal for each project (except Bonneville Dam where the goal was 85%) and a goal of 85% fish passage efficiency (FPE), where 85% of the juvenile salmon were to pass through non-turbine routes.

The Program and its amendments did not define exactly how to implement spill at each dam to achieve the survival and FPE goals. In the ensuing years, numerous technical discussions occurred between COE and the fish management agencies and tribes about the implementation of spill at each dam. The fishery agencies and tribes discussed spill and other fish passage operations with the COE prior to each fish passage season. These preseason discussions led to separate implementation plans, and the resultant in-season spill operation was a compromise between power production and fish passage, falling short of what the fishery agencies and tribes believed was necessary to meet the Program goals.

To end the dispute over spill required for passage, a 10-year Fish Spill Memorandum of Agreement (MOA) was signed on 10 April 1989 by the regional fishery agencies, Indian tribes, and the Bonneville Power Administration (BPA). This agreement was then adopted by the Council into the Program. The MOA provided a specific amount of spill at four mainstem Columbia River dams as "final and complete settlement of any claims regarding any obligation of the United States of America to provide spill as mitigation for mortality to anadromous fish attributable to turbine passage at all Federal Columbia River Hydroelectric Projects."

The four dams (The Dalles, John Day, Ice Harbor, and Lower Monumental) were included because they either did not have bypass systems or the existing bypass system was not performing to the expected standards. Implementation of this MOA was determined annually by the signatories and presented as the Fish Spill Implementation Plan in the Detailed Fishery Operating Plan published each year by the Columbia Basin Fish and Wildlife Authority. While not a signatory, the COE implemented the majority of the fishery agency spill requests at the dams included in the MOA. Bonneville Dam was not included in the MOA, and implementation of spill at this project required annual discussions.

In 1992, Snake River sockeye salmon (*O. nerka*) were listed by the National Marine Fisheries Service (NMFS-NOAA Fisheries) as endangered under the Endangered Species Act (ESA). In the following 6 years, additional Columbia Basin salmon stocks were listed under ESA as either threatened or endangered. With the listings came consultation between NOAA Fisheries and the action agencies and eventually a series of Biological Opinions (BiOps) which required specific spill scenarios at each FCRPS dam. Of particular importance were the 1995 and 1998 BiOps, which established the most extensive fish spill programs yet implemented in the Columbia Basin (NMFS 1995a, 1998b).

The 1995 FCRPS BiOp (NMFS 1995a) established a spill program to pass 80% of downstream migrants through non-turbine routes, or an FPE of 80%. This 80% FPE goal was to be achieved by spilling a specific percentage of total river flow at each FCRPS dam based on the performance of other fish passage devices already in place. For example, if a bypass system passed 50% of the daily fish passage, then spill was provided as necessary to pass an additional 30% of the fish. Specific spill hours and dates were prescribed for each dam along with spill limitations based on total dissolved gas generation and adult passage concerns.

In 1998, after new listings of steelhead stocks in the Columbia River Basin, the 1995 BiOp was supplemented with additional requirements to protect listed stocks (NMFS 1998b). This 1998 Supplemental Biological Opinion and its appendices required spill above the 80% FPE goal to aid system-wide juvenile fish passage by adding spill at projects where the 80% FPE goal could be exceeded without exceeding the dissolved gas limits, to make up for projects where spill was held below the 80% FPE goal by dissolved gas limits or other fish passage concerns. Spill levels specified in the 1998 Supplemental Biological Opinion can be found in Appendix A.

Present Status

The 2000 FCRPS Biological Opinion states "spill will be at the levels recommended in the 1998 Supplemental FCRPS Biological Opinion, assuming that waivers are obtained from the states of Oregon and Washington to exceed their 110% TDG state water quality standards. The Action Agencies would continue to provide spill for fish passage, but not to exceed TDG levels allowed under the standard or any modifications to it" (NMFS 2000).

Description of Spillways

The spillways of all FCRPS dams consist of a forebay, spill gates, ogee, stilling basin, and tailrace. The forebay typically consists of about 1 km of reservoir immediately above the spillway gates. Most spillway gates are a radial design with a 60-ft radius and 50-ft width (COE 1996b). Two dams (Bonneville and McNary) have vertically operated lift gates of similar width. Number of spillbay gates per spillway varies from 8 to 10 at lower Snake River dams to 18 to 23 at lower Columbia River dams. The ogee sections transition flow from below the gates to the stilling basin.

Flow deflectors that help reduce dissolved gas production are located on the ogee sections at elevations designed for each project. Below the ogee, the spilled flow enters a stilling basin designed to dissipate turbulent energy in a confined, armored zone, thereby minimizing the threat to the spillway's structure. Beyond the stilling basin, the tailrace extends for about 1 km downstream. Spillway capacities are designed for the maximum probable flood and vary from 850,000 cubic feet per second (cfs) at lower Snake River dams to 2,290,000 cfs at lower Columbia River dams

Spill Efficiency and Effectiveness

Spill effectiveness is the proportion of fish approaching a project that pass via the spillway. Spill efficiency is spill effectiveness divided by the proportion of total river flow passing over the spillway at the same time. Recent reviews of spill efficiency and effectiveness include Steig (1994), Giorgi (1996), Whitney et al. (1997), and Marmorek and Peters (1998). Estimates of spill efficiency vary by project.

Most spill efficiency estimates are based on hydroacoustic methodologies which make estimates for the general population, not specific species or stocks. Radiotelemetry studies have been conducted at some dams, where yearling and subyearling chinook salmon and steelhead were marked, released upstream, and their behavior and route of dam passage noted. With sufficient receiving antennae and sample size, estimates of spill efficiency and effectiveness can be derived.

Steig (1994) reviewed studies at Snake and Columbia River dams conducted through 1992 and noted that there is considerable variability in daily and weekly spill effectiveness. However, he concluded that most results fall around a 1:1 relationship between the proportion of water spilled and the proportion of fish passed in spill (i.e., 1.0 spill efficiency). Giorgi (1996) reviewed estimates of spill efficiency published through

1993 and noted that efficiencies are poorly estimated for most species due to a combination of sparse observations, imprecise estimates, and the reliance of most estimates on hydroacoustic monitoring which is unable to distinguish among species. He cautioned that the assumption of a spill efficiency of 1.0 could not be justified in most cases and implied that a suite of estimates acquired with different methodologies should be considered when attempting to derive species-specific estimates at individual dams. Relying largely on Giorgi's (1996) review, the PATH Hydro Work Group (PATH 1997) concluded that a range of spill efficiencies from 1.0 to 2.0 should be incorporated into sensitivity analyses of passage at Snake and Columbia River dams.

In contrast to all other dams, the powerhouse of The Dalles Dam is oriented nearly parallel to the natural course of the river, while the spillway is located on what was a shallow basalt bluff. The Independent Scientific Group (ISG 1996) suggests it is not surprising that this project exhibits higher spill efficiency than many other projects, due to its unique configuration. Giorgi and Stevenson (1995) reviewed biological investigations that described smolt passage behavior at The Dalles Dam and discussed implications to future surface bypass research. They cited three investigations that indicated that spill efficiency was near 2.0 when about 20% of the flow passed over the spillway. These included Willis (1982), which describes a curvilinear relationship in which spill efficiency is equal to or greater than 2.0 at spill below approximately 30% of total river flow, declining to about 1.4 at 60% spill, and further declining to 1.0 at 100% spill.

A recent radiotelemetry study at The Dalles by Holmberg et al. (1998) supports this general relationship. Spill efficiency for yearling chinook was 2.3 at 30% spill and 1.25 at 64% spill in 1996. PATH (1997) reviewed the available information and suggested that a factor of 2.0 be applied at The Dalles Dam at spill levels up to 30%; when spill levels are above 30% spill, the relationship grades from 2.0 to 1.0 according to Equation (1). This relationship predicts a spill efficiency of 1.5 at 65% spill:

$$
\begin{aligned}
P_f &= 2.0P_w & 0 < P_w \leq 0.30 \qquad (1)\\
P_f &= (2.43\text{-}1.43P_w)P_w & P_w > 0.30
\end{aligned}
$$

P_f is the proportion of fish passing over the spillway (spill effectiveness)
P_w is the proportion of total river flow passing over the spillway
Spill efficiency is defined as $P_f \div P_w$.

Spill efficiency at Lower Granite, Little Goose, and Lower Monumental Dams can be estimated based on radiotelemetry observations for yearling chinook salmon at Lower Granite Dam (Wilson et al. 1991), because of the similarity of the three projects.

Combining the radiotelemetry observations with the assumptions that 0% of fish pass the spillway at 0% spill and 100% pass at 100% spill, the following relationship (Smith et al. 1993) can be applied:

$$P_f = 2.583P_w - 3.250P_w^{\,2} + 1.667P_w^{\,3} \qquad (2)$$

The shape of this relationship is highly uncertain outside of the range of observations (20 to 40% spill), even though P_f must logically go to 0 and 1.0 at the extremes (ISG 1996).

Hydroacoustic studies conducted in 1997 and 1998 at John Day Dam (BioSonics Inc. 1999a, b) indicated high spill efficiency, particularly for daytime spill. For example, in the spring of 1998, daytime and nighttime spill efficiency was as high as 3.4 and 2.0, respectively. Summer efficiency in 1997 averaged 5.1 and 3.0 for daytime and nighttime spill, respectively. Daytime spill efficiency tended to decrease as spill percentage increased, whereas nighttime spill efficiency showed no significant correlation with spill level.

At John Day Dam in 1999 a constant nighttime spill of 60% was scheduled (but observed spill was actually 45%), and daytime spill was 0 and 30%, each scheduled for 3-d continuous periods within each replicate. Johnston and Nealson (1999) measured nighttime spill efficiencies during the spring at John Day Dam that were lower than reported previously, ranging from 1.4 to 1.7. During the summer 24-h spill efficiency ranged from 2.4 to 5.1, while nighttime spill efficiency ranged from 2.4 to 3.3. Both the spring and summer periods exhibited significantly higher 24-h spill effectiveness during the 30% daytime spill treatment.

Hansel and Beeman (1999) found steelhead spill effectiveness was not significantly different between the two treatments, at 46 and 54%, although effectiveness was higher during the treatment that included daytime spill. For yearling chinook salmon spill effectiveness was 53 and 65%; the periods when daytime spill was provided were significantly different and greater than the treatments without daytime spill. Both the hydroacoustic and radiotelemetry techniques are subject to sampling assumptions and error. In particular, hydroacoustic estimates are sensitive to transducer position, fish size, and fish orientation. During the summer, hydroacoustic estimates are particularly susceptible to error caused by the presence of non-salmonids, such as juvenile shad, and the high summer effectiveness values reported should be viewed with this in mind.

Spill efficiency and spill effectiveness were examined in 1999 at Ice Harbor Dam using radiotelemetry (Eppard et al. 2000) (note: definitions for spill efficiency and effectiveness have changed recently where efficiency = P_f and effectiveness = P_f / P_w). The study concluded that spill efficiency was 82.6% and spill effectiveness was 1.8:1 during BiOp day conditions (approximately 45% spill). During nighttime BiOp conditions (approximately 100% spill), spill efficiency increased to 96.1%, however, spill effectiveness was closer to a 1:1 relationship. This suggests that there may be an optimal level of spill at Ice Harbor Dam above which there is no gain in spill effectiveness at passing fish.

Seasonal Spill Timing

Historically, seasonal spill timing has been based on juvenile fish abundance. Early spill programs relied primarily on preseason planning dates and in-season estimates of cumulative fish passage as a trigger for the beginning and end of the spill season at each project. These programs were managed to provide spill for the middle 80% of the juvenile outmigration. These percentages were applied separately to the spring and summer migration periods and were based on actual fish sampling at each project via a smolt monitoring program (FPC 1995).

The NOAA Fisheries 1998 BiOp proposed that the actual dates of spill be determined annually by the regional Technical Management Team and based on in-season monitoring of abundance of tagged fish and population indices at Lower Granite and McNary Dams. The 1998 BiOp established Snake River spill planning dates as April 3 to June 20 and June 21 to August 31; lower Columbia River dates as April 20 to June 30 and July 1 to August 31, and mid-Columbia River dates as April 10 to June 30. One problem noted with this schedule is fish in the river after August 31 receive no benefit from the spill program. Under late-summer, low flow conditions, it takes subyearling chinook salmon approximately 3 weeks to travel from Lower Granite Dam to Bonneville Dam (COE 1999e).

Daily Spill Timing

The early spill programs based daily spill timing on hourly monitoring of smolt abundance at projects where this information was available. If hourly information was not available, daily spill timing was determined prior to each season and was based on the best available diel passage information. This diel information was obtained from smolt monitoring or research activities conducted primarily at powerhouses, rather than spillways, and it was common to apply diel information collected at one dam to another.

However, recent studies have shown that diel passage timing through spillways is different than powerhouses. For example, while smolt monitoring information suggests 60 to 90% of the daily powerhouse passage at John Day and Lower Monumental Dams occurs from 1800 to 0600 hours (Ransom and McFadden 1987, Martinson et al. 1997), hydroacoustic studies indicate that both day and night passage of juvenile migrants through the spillway is high at both dams during involuntary daytime spill (Johnson et al. 1998, BioSonics Inc. 1999a, b).

Observations at The Dalles and Bonneville Dams also indicate that spillway passage occurs at fairly constant rates, day and night, if daytime spill is provided (BioSonics Inc. 1997, Hensleigh et al. 1998). This suggests that juvenile migrations may be delayed if spill is managed based on powerhouse diel passage patterns. Studies conducted at John Day Dam in years with little daytime spill found that fish often milled in front of the dam when they arrived during the day and passed the dam at night, most often through the powerhouse (Giorgi et al. 1985, Sheer et al. 1997). In contrast, Sheer et al. (1997) fund that fish passing The Dalles Dam during 24-hour spill delayed very little, and most (85%) passed through the spillway. Hensleigh et al. (1999) and Liedtke et al. (1999) found a similar response at John Day Dam, where radio-tagged yearling and subyearling chinook salmon and steelhead had relatively short forebay residence times and high (50 to 75%) spillway passage rates during years where high flows caused high levels of daytime spill.

Forebay Predation

Beamesderfer and Rieman (1991) found that forebay populations of northern pikeminnow (*Ptychocheilus oregonensis*) and smallmouth bass (*Micropterus dolomieui*) were present in substantial numbers in the forebay of John Day Dam. Poe et al. (1991) reported that the diet of northern pikeminnow in the forebay of John Day Dam was 66% salmonid smolts. This suggests that delay of outmigrants in the forebay could reduce survival due to increased predation, and project operations such as daytime spill that decrease forebay residence time could increase survival.

Tailrace Passage

The concept of developing spill patterns at FCRPS dams specifically for fish passage was first addressed systematically in the 1960s to facilitate adult salmon passage into the adult fish collection systems. Junge (1967) observed improved adult salmonid passage under intermediate to large spill volumes if four or five gates at each end of the spillway were at low volume settings. At large dams this resulted in a tapered spill pattern near each end and a flat spill pattern across the central portion of the spillway. At smaller dams this produced a "crowned" pattern across the entire spillway tailrace, with the highest discharge in the middle bays.

He evaluated adult salmon passage success by comparing ladder passage counts associated with various spill patterns. The spill pattens he developed that appeared best for adult passage conflict with what is thought today to be best for juvenile passage (high shoreline velocities), since Junge kept near-shore velocities low to facilitate adult migration and passage into fishway entrances located along shorelines.

Smolt residence time in spillway tailraces is likely influenced by spill volume and pattern. High spill volume and water velocity push water and presumably juvenile salmonids out of the immediate tailrace, and help redistribute piscivorous predators (northern pikeminnow) away from the immediate spillway tailrace, reducing potential predation opportunities (Faler et al. 1988). Shively et al. (1996) found that ambient river flow velocities of at least 1 m/s were necessary to keep northern pikeminnow from holding in areas near bypass outfalls, and that the degree by which water velocity eliminated northern pikeminnow holding increased as outfall distance from shore and water depth increased.

Hansel et al. (1993) found that hydraulic cover such as eddies and backwaters at velocities below this threshold were preferred northern pikeminnow feeding habitats, particularly when near primary smolt outmigration paths. Spill patterns that facilitate rapid juvenile egress from the spillway stilling basin through the tailrace likely increase juvenile survival. Current spill patterns are developed to increase the survival of juvenile fish through tailraces, by emphasizing minimizing hydraulic cover and maintaining high water velocities near spillway shorelines. To not interfere with daytime adult passage, these juvenile spill patterns are often employed during nighttime hours only (COE 1999d).

As information is gained on the use and benefits associated with daytime spill to pass juveniles, greater consideration will likely be given to the use of juvenile spill patterns during the daytime. For example, in 2000 NOAA Fisheries proposed to evaluate spillway survival at The Dalles Dam using the juvenile (nighttime) spill pattern 24-hours/day (Dawley et al. 1999b). Spill patterns that attempt to satisfy both adult and juvenile passage criteria during the daytime have been developed for Bonneville and John Day Dams using scale hydraulic physical models at the COE Engineering Research and Development Center (ERDC).

The Bonneville and John Day Dam patterns have been implemented and juvenile and adult salmonid responses to these patterns are being evaluated through radiotelemetry studies. At John Day Dam, Liedtke et al. (1999) observed good juvenile egress through the spillway tailrace with the new spill pattern; marked yearling and subyearling fish passed through the first 0.7 km of tailrace in 5 to 10 minutes. They also observed slower passage times and higher predation rates on fish that passed through end spill bays.

Spill Survival

Whitney et al. (1997) reviewed 13 estimates of spill mortality for salmonids (3 steelhead and 10 salmon) published through 1995 and concluded that 0 to 2% is the most likely mortality range for standard spillbays. They also pointed out that local conditions, such as back eddies or other situations that may favor the presence of predators, may lead to higher spill mortality.

Some point estimates for mortality in spillbays with spill deflectors are higher than estimates for spillbays without deflectors. For example, the highest estimates of survival for yearling chinook salmon and steelhead at Snake River dams were obtained from spillbays without flow deflectors, ranging from 98.4 to 100% (Muir et al. 1995b, 1996, 1998). Although lower survival estimates were obtained from spillbays with flow defectors (ranging from 92.7 to 100%) (Iwamoto et al. 1994; Muir et al. 1995b, 1998), differences in survival between the two types of spillbays compared pairwise were not significant for steelhead at Little Goose Dam or yearling chinook salmon at Lower Monumental Dam.

Eppard et al. (2003) also reported that survivals were lower over spillways with flip lips when flows were lower during the spring of 2002 and the summers of 2000 and 2002. These authors suggested that the lower survivals may have been due to hydraulic conditions in the stilling basin when flows were less than 90 kcfs. In addition, for yearling chinook salmon at Lower Monumental Dam in 2003, relative survival increased by 4.0% when submergence of flip lips increased as flows increased (E. Hockersmith, NOAA Fisheries, Person. commun.). Using hydroacoustics at Ice Harbor Dam in 2003, Moursand et al. (2003) also found that route specific passage and project-wide performance metrics differed among operational treatments. Carlson and Duncan (2003) suggested that turbulence over the spillway ogee increased with decreased flow over the spillways at Ice Harbor Dam, causing a higher probability of fish contacting spillway surface structures during lower discharge. Flip lips were originally designed to reduce gas supersaturation when involuntary spill was required during periods of higher flow and subsequent higher tailrace elevation. Hence, survival over spillways with flip lips may be reduced somewhat when spill occurs at lower flow levels.

A number of methodologies have been used to estimate spillway survival at lower Columbia River dams, including identification of test fish by fin clips (Holmes 1952), freeze brands (Johnsen and Dawley 1974, Raymond and Sims 1980), coded-wire tags and freeze brands (Ledgerwood et al. 1990), balloon tags (Normandeau Associates Inc. et al. 1996a, b), PIT tags (Dawley et al. 1998b, 1999, 2000a, 2000b, Absolon et al. 2002), and radiotelemetry (Counihan et al. 2002).

At Bonneville Dam, Holmes (1952) estimated that subyearling chinook salmon survival through the spillway was 96 to 97%, depending on how the data were analyzed. Johnsen and Dawley (1974) compared the survival of subyearling chinook salmon passing through spillbays with and without flow deflectors, and found that relative survival was 87 and 96%, respectively, and that these differences were not statistically different. Ledgerwood et al. (1990) found that survival of subyearling chinook through spillbay 5 was not significantly different than for fish released downstream. Based on the balloon-tag methodology, the calculated survival probabilities for deflector and non-deflector spillways were both 1.0 at Bonneville Dam, however, fish passing through a spillbay without a spill deflector displayed a slightly higher injury rate (Normandeau et al. 1996a).

At The Dalles Dam, Dawley et al. (1998b) released PIT-tagged subyearling chinook and coho salmon in 1997, and estimated spillway survival of 87 and 92%, respectively, with 64% spill. Results from a 1998 study (Dawley et al. 1999a) show that relative survival rates during 64% spill were 88% for coho salmon and 76% for subyearling chinook salmon, while during 30% spill survival was 96 and 92% for coho and subyearling chinook salmon, respectively.

Preliminary analysis of data from 1999 show that relative survival rates during 64% spill were 94% for coho salmon and 95% for subyearling chinook salmon, while during 30% spill survival was 96 and 100% for coho and subyearling chinook salmon, respectively (Dawley et al. 2000b). In 2000, Absolon et al. (2002) estimated relative survivals at a 40% spill volume of 95% for yearling chinook and coho salmon and 92% for subyearling chinook salmon.

Estimates of spillway passage survival at John Day Dam are limited to a single study conducted in 1979 by Raymond and Sims (1980), who found that spillway mortality relative to the tailrace was not different from 0. PIT-tag estimates of spillway survival are presented in Table 1.

Survival of juvenile salmonids passing through spillways at lower Snake River dams has been evaluated, at least once, at all projects (Table 2). In 2000, NOAA Fisheries estimated relative survival for river-run hatchery yearling and subyearling chinook salmon (*O. tshawytscha*) passing through the spillway at Ice Harbor Dam on the Snake River. Relative spillway survival for hatchery yearling and subyearling chinook salmon, using PIT tags, was 0.978 (95% CI, 0.941-1.018) and 0.885 (95% CI, 0.856-0.915), respectively. The estimate for yearling chinook salmon was similar to spillway survival estimates at Little Goose (102.1%) (Iwamoto et al. 1994) and Lower Monumental Dams (92.7 to 98.4%) (Muir et al. 1995).

Estimated survival for subyearling chinook salmon passing through the spillways at lower Snake River dams was only evaluated one other year, at Lower Monumental Dam (Long et al. 1972). Survival estimates at Ice Harbor in 2000 were slightly higher than the estimates documented at Lower Monumental Dam (83.1 and 84.0%) and within the range reported at The Dalles Dam.

Table 1. Test conditions, number of replicates, number of PIT-tagged fish released, and survival estimates (standard error) through various routes of passage for turbine, bypass and spillway releases at Lower Granite (LGR), Little Goose (LGO), and Lower Monumental (LMO) Dams, 1993 to 1997.

Release location	Dam	Year	Species	Locations/ conditions	No. of replicates	No. of fish released (treatment/control)	Survival (s.e.)	Reference
Spillway	LGR	1996	Steelhead	Bay 1, 3.9 kcfs	5	7,491/7,468	1.010 (0.019)	Smith et al. 1998
(no	LGO	1997	Steelhead	Bay 1, 4.9 to 10 kcfs	14	6,736/6,953	1.004 (0.015)	Muir et al. 1998
deflector)	LMO	1994	Yearling chinook	Bay 8, 4.4 to 4.8 kcfs	3	4,157/4,243	0.984 (0.033)	Muir et al. 1995b
Spillway	LGO	1993	Yearling chinook	Bay 3, 3.8 kcfs	3	2,328/2,201	1.021 (0.026)	Iwamoto et al. 1994
(with	LGO	1997	Steelhead	Bay 3, 4.9 to 10 kcfs	15	7,494/7,453	0.972 (0.015)	Muir et al. 1998
deflector)	LMO	1994	Yearling chinook	Bay 7, 4.4 to 4.8 kcfs	3	4,206/4,243	0.927 (0.023)	Muir et al. 1995b
Bypass	LGR	1994	Yearling chinook	Unit 6A, Col. chan.	3	3,896/2,194	0.994 (0.030)	Muir et al. 1995a
	LGR	1995	Yearling chinook	Unit 6A, Col. chan.	4	3,130/3,021	0.976 (0.036)	Muir et al. 1996
	LGR	1995	Steelhead	Unit 6A, Col. chan.	5	3,747/3,763	0.983 (0.019)	Muir et al. 1996
	LGO	1994	Yearling chinook	Unit 6C, Col. chan.	3	3,407/2,225	0.994 (0.023)	Muir et al. 1995a

Table 1. Continued.

Release location	Dam	Year	Species	Locations/ conditions	No. of replicates	No. of fish released (treatment/control)	Survival (s.e.)	Reference
Bypass	LGO	1995	Steelhead	Unit 6C, Col. chan.	5	3,097/3,653	0.979 (0.031)	Muir et al. 1996
	LGO	1997	Steelhead	Unit 6B, Trashrack.	12	6,847/5,953	0.953 (0.016)	Muir et al. 1998
	LMO	1995	Yearling chinook	Unit 6C, Col. chan.	5	4,197/3,783	0.954 (0.034)	Muir et al. 1996
	LMO	1995	Steelhead	Unit 6C, Col. chan.	5	4,120/3,746	0.929 (0.060)	Muir et al. 1996
Turbine	LGR	1995	Yearling chinook	Unit 4B, 135 MW	2	3,236/1,581	0.927 (0.027)	Muir et al. 1996
	LGO	1993	Yearling chinook	Unit 6B, 135 MW	3	2,236/2,201	0.920 (0.025)	Iwamoto et al. 1994
	LGO	1997	Steelhead	Unit 6B, 135 MW	13	6,215/6,505	0.934 (0.016)	Muir et al. 1998
	LMO	1994	Yearling chinook	Unit 6B, 135 MW	2	2,838/2,841	0.865 (0.018)	Muir et al. 1995a

Table 2. Location, species and run type, study year, fish marking method, spillbay, test conditions, and survival estimates for spillway passage evaluation at hydroelectric projects on the lower Snake and Columbia Rivers. Abbreviations: LGR-Lower Granite Dam; LGO-Little Goose Dam; LMO-Lower Monumental Dam; IHR-Ice Harbor Dam; MCN-McNary Dam; JDD-John Day Dam; TDA-The Dalles Dam; BON-Bonneville Dam.

Dam	Species and run type	Year	Method	Flow deflector	Location	Conditions	Survival	Reference
LGR	Steelhead	1996	PIT tags	no	Bay 1	3.9 kcfs	1.01	Smith et al. 1998
LGO	Steelhead	1997	PIT tags	no	Bay 1	4.9-10.0 kcfs	1.004	Muir et al. 1998
LGO	Steelhead	1997	PIT tags	yes	Bay 3	4.9-10.0 kcfs	0.972	Muir et al. 1998
LGO	Yearling chinook salmon	1993	PIT tags	yes	Bay 3	3.8 kcfs	1.021	Iwamoto et al. 1994
LMO	Coho	1973	Freeze brands	yes*	Bay 2	4.5 kcfs	0.970	Long & Ossiander 1974
LMO	Coho	1973	Freeze brands	yes	Bay 4	4.5 kcfs	1.100	Long & Ossiander 1974
LMO	Steelhead	1974	Freeze brands	yes	Bay 7	4.5 kcfs	0.978	Long et al. 1975
LMO	Steelhead	1974	Freeze brands	no	Bay 8	4.5 kcfs	0.755	Long et al. 1975
LMO	Subyearling chinook salmon	1972	Freeze brands	yes*	Bay 2	13.1 kcfs	0.831	Long et al. 1972
LMO	Subyearling chinook salmon	1972	Freeze brands	yes*	Bay 2	2.8kcfs	0.840	Long et al. 1972
LMO	Yearling chinook salmon	1994	PIT tags	yes	Bay 7	4.4-4.8kcfs	0.927	Muir et al. 1995a
LMO	Yearling chinook salmon	1994	PIT tags	no	Bay 8	4.4-4.8 kcfs	0.984	Muir et al. 1995a
IHR	Yearling chinook salmon	2000	PIT tags	yes	Bays 3,5,7	BIOP night	0.978	Eppard et al. 2001
MCN	Subyearling chinook salmon	1955	Tattoo	no	not specified	not specified	0.980	Schoeneman et al. 1961

Table 2. Continued.

Dam	Species and run type	Year	Method	Flow deflector	Location	Conditions	Survival	Reference
MCN	Subyearling chinook salmon	1956	Tattoo	no	not specified	not specified	1.00	Schoeneman et al. 1961
JDD	Subyearling chinook salmon	1979	Freeze brands	no	Bay 16	4.3 kcfs	0.965-1.187	Raymond & Sims 1980
TDA	Coho	1997	PIT tags	no	Varied	64% spill	0.87	Dawley et al. 1998
TDA	Coho	1998	PIT tags	no	Varied	64% spill	0.89	Dawley et al. 2000a
TDA	Coho	1998	PIT tags	no	Varied	30% spill	0.97	Dawley et al. 2000a
TDA	Coho	1999	PIT tags	no	Varied	64% spill	0.93	Dawley et al. 2000b
TDA	Coho	1999	PIT tags	no	Varied	30% spill	0.96	Dawley et al. 2000b
TDA	Yearling chinook salmon	2000	PIT tags	no	Varied	BIOP	91.1	Absolon et al. 2002
TDA	Yearling chinook salmon	2000	Radio tags	no	Varied	BIOP	0.92	Counihan et al. 2002
TDA	Subyearling chinook salmon	1997	PIT tags	no	Varied	64% spill	0.92	Dawley et al. 1998
TDA	Subyearling chinook salmon	1998	PIT tags	no	Varied	64% spill	0.75	Dawley et al. 2000a
TDA	Subyearling chinook salmon	1998	PIT tags	no	Varied	30% spill	0.89	Dawley et al. 2000a
TDA	Subyearling chinook salmon	1999	PIT tags	no	Varied	64% spill	0.96	Dawley et al. 2000b
TDA	Subyearling chinook salmon	1999	PIT tags	no	Varied	30% spill	1.00	Dawley et al. 2000b
TDA	Subyearling chinook salmon	2000	PIT tags	no	Varied	BIOP	0.897	Absolon et al. 2002
TDA	Subyearling chinook salmon	2000	Radio tags	no	Varied	BIOP	0.826	Counihan et al. 2002
BON	Subyearling chinook salmon	1974	Freeze brands	no	Bay 11	13 kcfs	0.958	Johnsen & Dawley 1974
BON	Subyearling chinook salmon	1974	Freeze brands	yes	Bay 14	13 kcfs	0.868	Johnsen & Dawley 1974
BON	Subyearling chinook salmon	1989	CWT/Freeze brand	yes	Bay 5	6.8 kcfs	0.9604	Ledgerwood et al. 1990

*Flow deflector included dentates.

Dissolved Gas/Gas Bubble Disease

Dissolved Gas Standards

Recommended total dissolved gas standards for surface waterways were developed by the U.S. Environmental Protection Agency under the authority of the 1977 Clean Water Act amendment to the Federal Water Pollution Control Act of 1948. These standards were subsequently adopted by state environmental quality agencies. Each Pacific Northwest state has slightly different statutes.

However, in the case of the mainstem Snake and Columbia Rivers, a common standard of 110% total dissolved gas supersaturation (TDGS) was adopted. Each state has provisions for short-term modifications of the standard. In 1994, NOAA Fisheries first applied for and was granted a special spill program from Washington and Oregon. These modifications allowed total dissolved gas levels up to 115% in the forebay and 120% in the tailrace of each FCRPS dam. The 1995 FCRPS BiOp included these modified standards in the spill program requirements. Since 1994, NOAA Fisheries has requested and received annual water quality modifications from each state.

In 1997, the state of Washington replaced the dissolved gas standard annual modification with a "fish passage exemption" in the Water Quality Standards for Surface Waters of the State of Washington (Chapter 173-201A-060 (4)(b)), which mandates allowable dissolved gas limits identical to those in the NOAA Fisheries BiOp, with the addition of a 1-hour maximum of 125%. This exemption specifically states that it is intended for "spillage for fish passage" and is "temporary...to be reviewed in the year 2003." In qualifying for this exemption, NOAA Fisheries has met annual reporting and monitoring requirements.

Table 3 presents the estimated 1999 spill volumes which resulted in approximately 120% TDGS at tailwater gas monitors. Spill volumes vary with forebay TDGS, powerhouse flow volumes, and spillway tailwater elevations. Bonneville Dam spill levels are most often limited by the 115% dissolved gas limit at the Camas-Washougal, WA gas monitoring station, located downstream from Bonneville Dam.

Table 3. Estimated spill caps[a].

Project	Spill Caps (kcfs)
Lower Granite	55-65
Little Goose	40-50
Lower Monumental	35-45
Ice Harbor	95-105
McNary	120-135
John Day	75[b]-150
The Dalles	185-200
Bonneville	100-135

a Adapted from the COE of Engineers Fast-Track Gas Abatement Program (Rock Peters, COE, Portland District, pers. commun., May 1999).

b Gary Fredricks, NOAA Fisheries, pers. commun., March 2000.

Dissolved Gas Supersaturation

Soon after Bonneville Dam was completed in 1938, dead adult salmon were observed downstream by fishermen (Merrell et al. 1971). Unreconciled losses of adult salmon continued through the 1950s. Westgard (1964) first documented gas bubble disease (GBD) on the Columbia River in the McNary Dam spawning channel. Dissolved gas supersaturation was documented and unequivocally associated with spill at mainstem Columbia River dams in the late 1960s by Ebel (1969). By inference, Merrell et al. (1971) attributed earlier observed adult salmon losses to supersaturation. In 1967, run-of-the-river adult and juvenile salmon were observed with GBD and holding tests linked high dissolved gas and increases in temperature with mortality of juveniles (Ebel 1969).

Adult salmon mortality was documented in conjunction with high levels of supersaturation (125 to 135%) from Wells to Chief Joseph Dams from 1965 through 1969 (Meekin and Allen 1974). In 1968, when John Day Dam was completed but before turbines were in operation, dead adult chinook and sockeye salmon were found floating downstream and live fish with signs of GBD were captured (Beiningen and Ebel 1970). From 1968 to 1975, GBD in high-flow years contributed to high mortalities of juvenile salmonids migrating from the Snake River (Ebel et al. 1975).

During the late 1960s and early 1970s, a regional task force defined the problem and developed possible remedies for excessive spill, methods to diminish supersaturation, and strategies to ameliorate impacts of dissolved gas supersaturation to salmonids. The methods investigated and implemented that decreased spill and supersaturation were 1) to increase headwater storage to control flow during the spring freshet, 2) to install additional hydroelectric turbines at many dams, and 3) to install flow deflectors ("flip-lips") on spillway ogees at selected dams to reduce plunging and air entrainment of spilled water (Smith 1974). Additionally, various fish releases were made to coincide with decreased flow through the hydro-power system, and river water used in fish holding areas was degassed. As a result of these remedial measures, there was little evidence of GBD in salmonids in the late 1970s and 1980s (Dawley 1986).

Many studies on GBD and its effect on salmonids were conducted in an attempt to define the threat in the mainstem Columbia and Snake Rivers. The severity of GBD was dependent upon species, life stage, body size, level of total dissolved gas, duration of exposure, water temperature, general physical condition of the fish, and swimming depth (Ebel et al. 1975). A thorough review of the literature on dissolved gas supersaturation and of recorded cases of GBD was compiled by Weitkamp and Katz (1980) and updated by Fidler and Miller (1993). Despite numerous studies, there were still questions regarding the TDGS that migrating salmonids can safely tolerate and how to evaluate impacts. In 1994, NOAA Fisheries and BPA convened a panel of experts to review dissolved gas conditions on the river and assess impacts to salmonids from GBD resulting from voluntary spill (GBD Panel 1996). The panel concluded that not enough was known to accept the hypothesis that aquatic organisms in the Columbia and Snake Rivers were not impacted by dissolved gas from voluntary spill.

GBD Monitoring

Beginning in 1994, the annual TDGS waivers from the states were granted with the stipulations that proper monitoring would be conducted and results of research and monitoring would demonstrate minimal effects from GBD at temporarily increased levels of 115% in reservoirs and 120% in tailraces at dams. A formal plan was developed by NOAA Fisheries to monitor TDGS and signs of GBD in the aquatic biota (NMFS 1997). Critical uncertainties of the monitoring program were identified (Biological Monitoring Inspection Team 1995), gas bubble disease research priorities were developed (NMFS 1996), and thresholds of 15% prevalence and 5% severe GBD signs were set for continued voluntary spill.

Through the 1990s, TDGS monitoring improved. By 1998, timely data distribution by COE included hourly dissolved gas levels and water temperatures from 41 monitoring sites in forebays and tailraces on the mainstem Columbia and Snake Rivers (COE 1998b). Quality assurance and control measures and improved gas measurement technology increased data precision and provided confidence that fisheries management decisions regarding the threat from TDGS were based on sound data. However, concerns still existed over whether the monitoring sites appropriately represent all conditions experienced by salmonids (ISAB 1999). Additionally, intensive monitoring throughout reservoirs and in tailraces has allowed a series of models to be developed that relate tailwater TDGS to spill, total project release, and forebay TDG for all mainstem dams from Grand Coulee and Lower Granite to Bonneville Dams, as well as from Dworshak Dam (COE 1998a).

Through the 1990s, GBD monitoring of migrating juvenile and adult salmonids was also greatly improved. Numbers of fish, methods, locations, and times for sampling were adjusted to provide a representative network of samples. Quality assurance and control methods met regional requirements, thus providing confidence that fisheries management decisions were based on the best possible data given the present state of technology.

Uncertainties associated with the GBD monitoring are 1) whether the relationship between smolt/adult mortality and gas bubbles in fins, gills, and lateral line is known, 2) whether clinical signs change during collection and examination, 3) whether signs in sampled fish represent the river site over the entire 24-hour period, 4) whether samples are taken at representative locations, including those of high risk from GBD, 5) whether sample size is statistically adequate for required confidence limits, 6) whether key signs of GBD and their relative significance are known, and 7) whether the 15% threshold level for GBD prevalence can be tolerated by the juvenile migrant population. Research conducted to address these uncertainties suggests:

1) The relationship between prevalence and severity of GBD signs and mortality of juvenile salmonids is unresolved. Research has produced an equivocal relationship except at consistent 130% TDGS (Mesa et al. in press).

2) Varying pressures encountered by juvenile salmonids during passage through the collection and bypass systems at dams appears not to substantially alter existing GBD signs nor induce new signs (Absolon et al. 1999). Collection protocols require fish be collected at the dewatering screens in the bypass conduit, held in river water at ambient temperature, and processed within 15 minutes of sampling (Maule et al. 1997). Prevalence of GBD signs in fish collected from bypass systems was somewhat greater than that of fish collected by seines from reservoirs (Backman et al., in prep.). The increased GBD signs may be caused by passage delays in front of dams prior to bypass system entry. The ISAB (1999) called for more research to evaluate this difference.

3) Fish samples collected for GBD assessment of smolt monitoring facilities at Snake and Columbia River dams adequately represented fish passing through the smolt bypass systems 24 hours per day (NMFS 1999b).

4) Smolt monitoring sites appeared to provide samples that were generally representative of upstream and downstream locations.

5) Evaluation of sample statistics by Maule et al. (1997) showed that numbers of sampled fish were generally sufficient to document a species-specific GBD prevalence when impacted migrants reached the NOAA Fisheries threshold of 15% prevalence or 5% severe signs of GBD. An exception is at McNary Dam where migrants from the Snake River cannot be differentiated from Columbia River migrants. Here, a low prevalence in the total sample could be associated with a large prevalence within the Snake River subpopulation. However, at

dissolved gas levels present during voluntary spill, this problem is insignificant.

6) The GBD signs utilized as a GBD index for smolt monitoring are externally visible subcutaneous emphysema. Research to identify other indicators of GBD such as blood chemistry changes was not pursued. Acoustic assessment of internal gas emboli was determined too expensive and not suitable for use in field locations (Carlson 1995). Excision of gill lamella was investigated, but thought to be no better an index of GBD impacts than other less invasive methods.

7) Signs of GBD can only be utilized as a subjective tool. The GBD Panel (1996) stated that when GBD signs are observed in any fish, there should be regional concern about survival. Also, the greater the prevalence and severity of GBD signs the more concerned managers should be about fish survival.

GBD Impacts

Results of GBD monitoring and research on juvenile salmonids in the 1990s show that incidents of high prevalence and severity of GBD signs and probable related mortality are associated with exceptionally high river flows and/or exceptional dam operation problems where powerhouse flows were limited. In 1990, a fire at John Day Dam caused 100% spill for an extended period, resulting in downstream TDGS levels greater than 130% and GBD prevalence up to 74% in steelhead and 38% in coho salmon.

In 1993, turbine outages at Lower Granite Dam caused TDGS to increase resulting in 18% prevalence of GBD in migrants at Little Goose Dam. In 1995 and 1996, turbine outages at Ice Harbor Dam coupled with high river flows caused TDGS exceeding 130% during the spring freshet. PIT-tag interrogation data collected at Snake and Columbia River dams suggested losses of juvenile chinook salmon coincident with the high levels of TDGS in the 67-km reach from Lower Monumental Dam to the Snake River mouth in 1995 (Cramer 1996a) and 1996 (Cramer 1996b, NMFS 1997). However, no losses of steelhead were measured in that same river reach for fish that prior to release at Little Goose Dam had been experimentally exposed to supersaturation sufficient to produce low-grade mortality (Monk et al. 1997a). Resident fish downstream from Ice Harbor Dam also showed high prevalence and severity of GBD signs in 1995 and 1996 (Schrank et al. 1997, 1998).

High river flows in 1996 and 1997 caused excessive spill at all mainstem dams, and TDGS exceeded 120% (the maximum allowed for voluntary spill) for extended periods throughout most of the lower Snake River and mid- and lower-Columbia River.

Results of GBD monitoring showed that prevalence of subcutaneous emphysema in juvenile salmonids was minor when TDGS was 120 to 125%, but was generally 10% or greater when TDGS was higher than 125% (NMFS 1998a). Under high river flow conditions, even when all possible system management actions are taken, TDG often cannot be kept below 120%.

In the early 1990s, river flow was low and often within powerhouse hydraulic capacities. When there was sufficient market for power, voluntary spill for fish passage and TDGS at 120% in tailraces and 115% in forebays could be regulated. Effects of GBD under these conditions appeared benign, signs were minimal or non-existent, and there was no apparent GBD related mortality (Maule et al. 1997, ISAB 1999). Even during periods of involuntary spill, GBD impacts appeared to be minor, except when TDGS was over 120% (NMFS 1997, 1998a, 1999b).

Impacts of GBD as measured by visual external signs in river migrants are greater at higher temperatures, particularly when the ambient water temperature of supersaturated water increases (Ebel 1969, Ebel et al. 1971). Steelhead have a lower tolerance to TDGS than other species based on Smolt Monitoring Program data (NMFS 1997, 1998a, and 1999b) and laboratory tests (Fidler and Miller 1993).

Dissolved Gas Abatement

Starting in 1994, a COE Dissolved Gas Abatement Study developed concepts for decreasing TDGS at the eight Snake and Columbia River dams. The COE is pursuing numerous potential structural modifications to spillways and stilling basins for dissolved gas abatement alternatives. While some of these alternatives may be implemented in the future, only flow deflectors have been implemented to date to reduce gas production at the FCRPS projects.

None of the gas reduction alternatives evaluated met biological, operational, and economic criteria, nor did they decrease gas levels to the 110% water quality standard for the 7-day 10-year discharge events. Whitney et al. (1997) found that under appropriate operating conditions, flow deflectors generally lowered dissolved gas levels downstream from the projects by 10 to 20% for a given spill flow. Deflectors installed recently at Ice Harbor and John Day Dams have provided gas abatement benefits in the upper portion of this range.

JUVENILE PASSAGE THROUGH MECHANICAL SCREEN BYPASS SYSTEMS

Description of Systems

Two submersible fish screen designs are used at Columbia River hydroelectric dams to guide fish away from turbine intakes and into juvenile bypass systems: a submersible traveling screen (STS) and an extended submersible bar screen (ESBS). STSs have a monofilament mesh screen that rotates around two large rollers at the top and bottom of the screen. The screen is rotated periodically to allow flow passing through the screen to flush the mesh surface clean of debris. STSs are currently installed at Lower Monumental, Ice Harbor, John Day, and Bonneville Dams. ESBSs are made of a fixed wedgewire screen material and have a bar sweep that is turned on periodically to brush debris from the face of the screen. ESBSs are currently installed at Lower Granite, Little Goose, and McNary Dams. The Dalles Dam does not have a mechanical screen juvenile bypass system.

STSs and ESBSs guide migrating juvenile salmonids from turbine intakes into gatewells. Fish exit the gatewells through orifices and enter a collection channel that travels the length of the powerhouse. The channel conveys fish and orifice flow from all gatewells directly to the river or to dewatering facilities. The dewatering facilities reduce flow to approximately 30 cfs, and flumes transport fish and the 30 cfs flow to a tailrace bypass outfall or barges or trucks for transportation. Smolt monitoring facilities installed at key bypass systems allow species composition, fish condition, run timing, and passage indices to be estimated. PIT-tagged fish can be detected at these facilities, time and date of passage noted, and fish diverted for further evaluation, if required.

Mechanical screen bypass system design criteria are described in NOAA Fisheries Juvenile Fish Screen Criteria (NMFS 1995b), COE bypass system design memorandums (COE 1995a, 1996a, 1999a), the COE Fish Passage Plan (COE 1999d), and an intake design guidelines manual (ASCE 1995). NOAA Fisheries guidelines for locating and designing bypass outfalls are presented in NMFS (1995b).

Fish Guidance Efficiency

Fish guidance efficiency (FGE) is a measure of how efficiently turbine intake screens guide juvenile salmonids out of turbine intakes. FGE is calculated as gatewell catch (guided fish) divided by the total number of fish (guided plus unguided) passing through the turbine intake during the test period (Brege et al. 1992). Fyke-nets are most commonly used to sample unguided juvenile salmonids. Gatewell dip-netting provides estimates of guided fish (Swan et al. 1979). Gatewell dip-net recapture efficiency tests with yearling and subyearling chinook salmon produced recapture efficiencies of 95 to 100% (McComas et al. 1994, Brege et al. 1997a,b, 1998).

Guidance screens are deployed and operated from late March through late fall or early winter, depending on the project (COE 1999d). Fyke-net FGE evaluations target specific species or life histories, usually yearling and subyearling chinook salmon, and steelhead. Incidental catches of sockeye and coho salmon are reported if numbers captured during the test are statistically meaningful. Tests are conducted during the main portion of the outmigration and evening hours to obtain sufficient sample sizes. Results are reported as daily values and seasonal averages. The results represent values for the entire population, which is a mixture of wild and hatchery fish and all stocks.

Early FGE studies utilized fyke-net frames beneath STSs to collect unguided fish (Krcma et al. 1986). Beginning in 1993, streamlined fyke-net frames and nets were designed to test extended screens, and placed in gatewell slots located downstream from the guidance screen (Brege et al. 1994). The newly designed fyke nets and frame and placement downstream from the guidance screen minimized water resistance and possible effects on FGE.

Williams et al. (1996) compared PIT-tag detection with fyke net estimates of FGE at McNary Dam and suggest a pressure field created by the fyke-nets located under the STSs biased yearling chinook FGE estimates upward. PATH (1998) developed a correction factor to adjust STS FGE estimates based on fyke-net position to improve the accuracy of the PATH retrospective model analysis for yearling chinook salmon. The correction factor was the ratio of FGEs derived at McNary Dam with the nets in the downstream slot compared to the upstream slot (i.e., directly beneath the STS). Kransow (1998) estimates the correction factor to be 0.82. The validity of applying the FGE correction factor for yearling chinook salmon at McNary Dam to other intakes and species is unknown. For example, McNary Dam has a large, low velocity intake compared to most other powerhouse intakes. Monk et al. (1999a) acknowledged the PATH FGE adjustment but chose not to apply it to the data for Bonneville Second Powerhouse. In their view, the correction factor was not considered applicable to ESBS FGE at Bonneville Second Powerhouse because the intakes at McNary Dam and Bonneville Second Powerhouse are hydraulically different, nor would the correction have affected the conclusions of their analysis.

Recently, FGE estimates have been obtained from the probability of detecting PIT-tagged fish (Smith 1997, Kransow 1998, Anderson et al. 1998), and radiotelemetry and hydroaoustic studies. Differences exist among the FGE estimates derived from these methodologies, and each method has associated sampling strengths, weaknesses, and critical assumptions. For example, when using the probability of detecting PIT-tagged fish to estimate FGE, results are most reliable when groups of tagged fish pass during periods of near 0% spill. When groups pass during periods of relatively high spill the method produces results dependent on the questionable assumption that spill index and detection probability have the same linear relationship throughout the range of observed spill values (Smith 1997).

FGE estimates vary with sampling method, species, rearing history, stock, fish condition (disease and smoltification), project, time of day, day, season, turbine unit, fyke net location, environmental conditions, and project operations. This spatial and temporal variability is likely related to complex interactions between biological and physical factors, including the arrival of different stocks at a dam throughout the season. Williams et al. (1996) attempted to clarify some of the mechanisms affecting FGE. However, FGE data based on fyke net estimates is generally limited to studies designed to compare FGE under different (prototype and hydraulic) test conditions, and little could be concluded. Gessel et al. (1991) suggest that factors such as water temperature, turbidity, flow, photoperiod, physiological development, and predation by northern pikeminnow may affect FGE.

This inherent variability makes it difficult to develop accurate and precise single-point estimates of FGE, and to use typical scientific format that would express FGE as a mean with a standard deviation. Even with this variability, single-point estimates of FGE are often used for modeling or management purposes. Table 4 presents values commonly used by PATH (Anderson et al. 1998) and NOAA Fisheries (Krasnow 1998) in modeling analyses. They reviewed the available data and developed point estimates of FGE for each project and certain species and rearing history.

NOAA Fisheries developed a spreadsheet model (SIMPAS) to evaluate benefits to fish survival associated with various FCRPS alternatives (Gary Fredricks, NOAA Fisheries, Pers. commun., August 1999). FGE values used in SIMPAS differ slightly from Krasnow (1998). No single estimate of FGE for each species and dam is universally accepted, and others in addition to those presented in Table 4 have been developed. To characterize the variability in the data, Table 5 also presents ranges in the actual daily FGEs measured through various sampling methodologies. Point estimates of FGE that are used to model the hydropower system should fall within these ranges.

Despite the variability discussed above, some general FGE trends have been observed. Side-by-side fyke-net estimates of FGE with STSs and ESBSs generally indicate that FGE is statistically significantly higher with ESBSs (McComas et al. 1993, Brege et al. 1994). FGE for yearling chinook salmon at Lower Granite Dam generally increases over time from throughout the migration season (Swan et al. 1990). A positive correlation was found between FGE and the level of smolt development exhibited by the migrant population (Giorgi et al. 1988).

With subyearling chinook salmon an opposite trend occurs; FGE generally decreases as the season progresses (Brege et al. 1988, Monk et al. 1999a). In the lower Columbia River, FGE of coho salmon is generally high and similar to steelhead, and sockeye FGE is generally lower than that of all other yearling migrants. FGE has been increased by a variety of adaptations to the basic STS, including extending the length of the screen from 20 to 40 feet (STS versus ESBS), raising the operating gate, lowering STS screens to open the throat area, and the use of an inlet flow vane with ESBSs to direct more flow into gatewell slots.

Table 4. Fish guidance efficiency (FGE) at Columbia and Snake River dams for 1999 configuration.[d]

Site (Screen type)	Species	PATH FGE (%)[a]	NMFS FGE (%)[b]
Lower Granite Dam (ESBS)	Yearling chinook	78	78
	Subyearling chinook	-	53
	Steelhead	-	81
Little Goose Dam (ESBS)	Yearling chinook	82	82
	Subyearling chinook	-	45
	Steelhead	-	81
Lower Monumental Dam (STS)			
	Yearling chinook	61	61
	Subyearling chinook	-	49
	Steelhead	-	82
Ice Harbor Dam (STS)	Yearling chinook	71	71
	Subyearling chinook	-	46
	Steelhead	-	93
McNary Dam (ESBS)	Yearling chinook	95	95
	Subyearling chinook	-	62
	Steelhead	-	89
John Day Dam (STS)	Yearling chinook	67	64
	Subyearling chinook	-	34
	Steelhead	-	85
The Dalles Dam (None) [c]	Yearling chinook	46	46
	Subyearling chinook	-	46
	Steelhead	-	40
Bonneville Dam			
Powerhouse One (STS)			
	Yearling chinook	41	38
	Subyearling chinook	-	16
	Steelhead	-	41
Powerhouse Two (STS)			
	Yearling chinook	43	44
	Subyearling chinook	-	18
	Steelhead	-	48

a) Based on report to PATH from Anderson et al. (1998).

b) Based on NMFS sensitivity run #1 (assumes $FGE_{ESBS}>FGE_{STS}$ for wild yearling chinook salmon).

c) FGE values for The Dalles are based on passage through the ice and trash sluiceway.

d) These estimates likely have range, but that range changes with a number of factors and is not easily estimated.

Table 5. Daily estimates of Fish Guidance Efficiency (FGE) for existing conditions. Hydroacoustic estimates are not species or rearing history specific. Estimates come from a variety of sources, and include adjustments for fyke net position to yearling chinook salmon FGE with STSs.

	Species	Fyke net		Hydro-acoustics		Radio-telemetry	
Project (Screen type)		**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Lower Granite (ESBS)	Yearling chinook			68[a]	93[a]	56[b]	62[b]
	Steelhead			68[a]	93[a]	84[b,c]	86[b,c]
	Subyearling chinook					42[b]	53[b]
Little Goose (ESBS)	Yearling chinook	67[d]	87[d]	17[e]	100[e]		
	Steelhead	88[d]	95[d]	61[e]	100[e]		
	Subyearling chinook						
Lower Monumental (STS)	Yearling chinook	43f	83[f]				
	Steelhead	58[f]	95[f]				
	Subyearling chinook	30[g]	40[g]				
Ice Harbor (STS)	Yearling chinook	53[h]	85[h]			50[i]	100[i]
	Steelhead	65[h]	95[h]				
	Subyearling chinook						
McNary (ESBS)	Yearling chinook	70[j]	96[j]	13[e]	100[e]		
	Steelhead	75[j]	97[j]	12[e]	100[e]		
	Subyearling chinook	45[j]	86[j]				
John Day (STS)	Yearling chinook	54[k]	78[k]			53[l]	84[l]
	Steelhead	65[k]	87[k]			67[l]	86[l]
	Subyearling chinook	13[k]	55[k]				

Table 5. Continued.

Project (Screen type)	Species	Fyke net Min	Fyke net Max	Hydro -acoustics Min	Hydro -acoustics Max	Radio -telemetry Min	Radio -telemetry Max
The Dalles	NA						
Bonneville Dam -First Powerhouse	Yearling chinook	21[m]	65[m]	15[n]	100[n]		
	Steelhead	27[m]	87[m]	15[n]	100[n]		
	Subyearling chinook	2[m]	43[m]	0[n]	100[n]		
Bonneville Dam -Second Powerhouse	Yearling chinook	22[m]	65[m]	0[n]	100[n]	29[o]	30[o]
	Steelhead	15[m]	70[m]	0[n]	100[n]	51[o]	52[o]
	Subyearling chinook	19[m]	62[m]	0[n]	100[n]		

a Johnson et al. (1999)
b Adams et al (1998)
c Includes ranges estimates for wild and hatchery steelhead
d Gessel et al. (1995)
e Anonymous (1996)
f Ledgerwood et al. (1987) and Gessel et al. (1993); range includes FGEs adjusted for fyke net frame position per Krasnow (1998)
g Ledgerwood et al. (1987)
h Brege et al. (1988); range includes FGEs adjusted for fyke net frame position and raised operating gate per Krasnow (1998)
i Pers. commun., Brad Eppard, NOAA Fisheries, February 2000
j McComas et al. (1995)
k Krcma et al. (1986) and Brege et al. (1987); range includes yearling migrant FGEs adjusted for fyke net frame position per Kransow (1998)
l Hansel et al. (1999a)
m Bonneville First: Monk et al. (1992a, 1993); Bonneville Second: Monk et al. (1994, 1995)
n Ploskey et al. (1998)
o Hansel et al. (1999b)

Orifice Passage Efficiency

Orifice passage efficiency (OPE) is the percentage of guided juvenile salmonids which exit the gatewell via the orifice in a given time period (usually 24 hours). The most recent estimates of OPE conducted at Lower Granite, Little Goose, McNary, and John Day Dams and at Bonneville First Powerhouse have been conducted by releasing either fin-clipped or PIT-tagged fish into the gatewell with the orifice open. With the fin-clipped fish, after 24 hours, any remaining fin-clipped fish are removed from the gatewell and counted. The percent that left the gatewell in 24 h is the percent OPE. With the PIT-tagged fish, the number exiting the gatewell is recorded by PIT-tag detectors in the bypass system.

The regional accepted minimum level for OPE with STSs installed is 70%. However, because of the increased flows and higher turbulence in the gatewell associated with ESBSs, OPE levels approaching 90% are probably more appropriate for gatewells with ESBSs. At Columbia and Snake River dams where OPE has been estimated, either with ESBSs or STSs in place, estimates have generally been greater than 70% for yearling and subyearling chinook salmon and steelhead, with the exception of McNary Dam which had lower OPE estimates with ESBSs with much greater variability (Brege et al. 1998) (Table 6). Evaluations of OPE for coho and sockeye have not been conducted.

Table 6. Most recent orifice passage efficiency (OPE) estimates for existing conditions (and prototype testing) at Snake and Columbia River dams.

Project	Species	OPE[a] (%)	Reference
Lower Granite	Yr. Chinook	95	Monk et al. 1997b
ESBS, 25-cm orifice	Steelhead	97	
Little Goose	Yr. Chinook	97	Gessel et al. 1996
ESBS, 25-cm orifice			
Lower Monumental[a]	Yr. Chinook	88	Gessel et al. 1993
STS, 30-cm orifice	Steelhead	66	
Mc Nary	Yr. Chinook	69	Brege et al. 1998
ESBS, 30-cm orifice	Subyr. Chinook	79	
John Day	Yr. Chinook	81	Brege et al. 1997a
Existing STS, 35-cm orifice	Subyr. Chinook	98	
John Day	Yr. Chinook	99	Brege et al. 1997a
Prototype ESBS, 35-cm orifice	Subyr. Chinook	97	
Bonneville Dam 1st Powerhouse	Yr. Chinook	80	Monk et al. 1999b
Existing STS, 30-cm orifice	Subyr. Chinook	98	
Bonneville Dam 1st Powerhouse	Yr. Chinook	90	Monk et al. 1999b
Prototype ESBS, 30-cm orifice	Subyr. Chinook	97	
Bonneville Dam 2nd Powerhouse[b]	Yr. Chinook	97	Monk et al. 2002
Modified screen system	Yr. Chinook	94	
Bonneville Dam 2nd Powerhouse[b]	Subyr. Chinook	100	Monk et al. 2002
Modified screen system	Subyr. Chinook	99	
Existing STS, 30-cm orifice			

a All estimates (except Lower Monumental) done with releases of fin-clipped fish; Lower Granite estimates are results of PIT-tagged and fin-clipped releases averaged together.

b Estimates made using PIT-tagged fish.

Separators and Separation Efficiency

Separation of outmigrant juvenile salmonids by size is an integral part of the juvenile bypass programs at FCRPS dams. Separators at Columbia River juvenile fish bypass facilities sort fish using an array of appropriately spaced bars oriented parallel to flow passing through a tank. This allows smaller fish to pass between the bars while excluding larger fish. The strategy for using bars for separation falls into two categories, single- and two-stage, depending on the objective of the process. The Dalles Dam has no juvenile fish bypass system or separator.

Single-stage separators are in use at Ice Harbor, John Day, and Bonneville Dams, where the primary purpose of separation is to monitor the condition of juvenile salmon and steelhead passing through the project bypass system. These units remove adult salmonids and large incidental species using "dry" separation in which smaller fish (including salmonid smolts) fall between separation bars along with incident flows, while large animals are carried across a short section of exposed bars into another flume for eventual return to the river. Smolts can then be sampled without concern for injury induced by larger adult and incidental fish in the sample holding area. At John Day Dam, the adult fish separation bars are wetted by small low-pressure jets along their upper surface to facilitate fish movement along the bars. Following the separation and sampling process, facilities using single-stage separators bypass all fish to the river downstream from the dam.

Two-stage separators used at McNary, Lower Monumental, and Little Goose Dams are intended to separate large steelhead from smaller chinook salmon smolts, as well as removing adult salmonids and large incidentals, prior to collection for transport using either barges or trucks. Size separation also allows selective bypass or transport of one or both smolt size-classes.

The separators at these three sites rely on 'wet' separation by keeping the fish submerged throughout the process. Each separator unit consists of three consecutive chambers (McComas et al. 1998). The first two chambers have separation-bar arrays with bars spaced approximately 19 mm (0.75 in) and 32 mm apart in the upstream and center chambers, respectively. Individual bars are 32-mm aluminum tubing. In both compartments, the arrays are submerged and have a slight positive slope, so that water depth over the array at the upstream end of each chamber is approximately 50 mm (2 in) deeper than at the downstream end. Water depth in the unit is controlled primarily by dewatering transport flow immediately upstream from the first chamber. Secondary control is provided by inflow points through perforated plate false bottoms in the first two sections. The most downstream (third) chamber is a simple box with no separation-bar array, water control structures, or false bottom.

Wet separators depend on sounding behavior, as an avoidance response to shallow conditions above the separation bars, to achieve separation (Gessel et al. 1985). All fish are introduced to the upstream compartment along with about 0.06 m^3/s transport flow from the bypass channel (Katz et al 1999). The smaller fish are filtered between bars in the upstream compartment, while larger smolts are removed in the second section. Large incidentals and adults pass through the unit to the third compartment for return to the river.

By keeping the animals submerged, wet separation is considered less stressful to fish. However, separation efficiency of operational wet separators is usually less than 70% for small fish, and varies considerably from year to year (Table 7). In addition, recent behavior and physiology studies have indicated that fish hold under the bars for extended periods rather than exit expeditiously from the wet separator unit (James L. Congleton, Idaho Cooperative Fish and Wildlife Research Unit, Department of Fish and Wildlife, University of Idaho, Moscow, Idaho 83844-1141, Pers. commun., 21 March 95). This suggests that many fish exit only after becoming fatigued as a result of swimming to avoid hydraulic conditions within the unit.

The separator at Lower Granite Dam is unique, in that a single-stage wet separation process is employed to segregate large incidental and adult salmonids from smaller fish. Presently, smolts are not separated by size. This system is scheduled for modification when with the entire Lower Granite Dam juvenile fish bypass facility is upgraded.

A research program was initiated in 1996 to evaluate potential improvements to existing wet separators, and to develop a new separator concept (Katz 1999, McComas et al. 2001a). These studies evaluated the effects of physical separator design including separation bar length, slope, color, and spacing as well as effects of transport flow, water velocities through the separator, depth over separation bars, and exit orifice orientation and configuration (McComas et al 1998, McComas et al 2000, McComas et al 2001b). Other work considered parameters independent of physical design which may affect separation potential, such as incident light and fish density (McComas et al. in prep a (2000 data), McComas et al. in prep b (2001 McN)).

Design criteria from those research efforts resulting in positive affects on separation efficiency and separator exit efficiency were incorporated into a separator insert used for comparison to the operational separator at McNary Dam (McComas et al. In prep b). The insert was an open aluminum box constructed of 48-mm (3/16-in) aluminum plate and sized to fit tightly within the upstream section of the operational McNary separator when the operational separation bars were removed. In addition to enhancements to increase separation efficiency (separation-bar spacing, slope, and submergence), the insert included a perforated-plate false bottom to reduce volume available for residence under the separation bars, and a more efficient submerged exit-orifice configuration.

Evaluation of the insert against the McNary operational separator over a series of 2-d replicates resulted in measurably higher separation efficiency values using the insert than for the McNary operational separator condition, though the difference was only significant for large fish groups and the total salmonid catch (Table 8). Using lights above the separator enhanced separation efficiency for all small salmonid groups although the relationship was significant only for the total small salmonid catch and for the total salmonid catch.

Table 7. Annual separation efficiency (%) by species for outmigrant salmonid smolts using operational two-stage wet separators at Federal Columbia River Power System (FCRPS) dams on the Columbia and Lower Snake Rivers, 1994-2003.

	Annual separation efficiency (%)										
Species	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	Reference
McNary Dam Juvenile fish bypass facility											
yearling chinook salmon	35.5	52.8	36.8	39.3	41.4	46.4	46.4	61.4	68.6	61.3	1994-1998[a] 1999-2003[b]
subyearling chinook salmon	48.7	58.1	46.3	56.9	49.8	50.8	53.4	66.0	64.3	59.8	
steelhead, hatchery	96.1	84.6	88.1	75.5	84.1	83.0	85.9	81.5	81.9	82.6	
steelhead, wild	80.3	55.1	67.0	59.1	62.8	62.9	73.6	63.4	60.6	65.5	
sockeye salmon	27.4	35.1	22.7	34.5	26.7	16.5	22.7	48.3	35.4	25.8	
coho salmon	24.3	20.6	25.6	38.1	22.9	18.8	23.9	27.9	30.9	28.8	
Lower Monumental Dam Juvenile fish bypass facility											
yearling chinook salmon, hatchery	65.6	61.6	42.2	63.9	50.9	45.4	36.2	52.3	87.1	70.1	1994-1998[c] 1999-2003[d]
yearling chinook salmon, wild	57.5	56.1	38.7	49.5	39.2	36.6	35.8	52.2	86.5	72.9	
subyearling chinook salmon	31.3	17.8	18.8	22.2	15.7	43.1	45.1	52.2	34.3	59.2	
steelhead, hatchery	65.1	65.2	66.2	55.6	72.7	65.2	56.5	71.3	75.2	72.2	

[a] Hurson et al. 1999.

[b] Rosanna Tudor, Washington Department of Fish and Wildlife, pers. commun., November 2003.

[c] Spurgeon et al. 1998.

[d] An experimental separator insert has been installed and in use in the small fish (upstream, 'A') section since 2001 (2002-2003 outmigration years).

Table 7. Continued.

Species	Annual separation efficiency (%) 1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	Reference
steelhead, wild	40.0	27.4	45.8	34.7	50.1	39.5	70.5	56.8	47.0	43.9	
sockeye salmon, hatchery	- [e]	4.6	10.8	11.1	15.1	17.1	27.0	50.1	63.8	78.1	
sockeye salmon, wild	24.4	19.4	21.9	24.9	7.1	28.8	34.1	6.3	41.8	60.0	
yearling chinook salmon, hatchery	47.6	60.3	70.9	60.0	64.8	78.5	77.8	64.6	60.4	71.2	1994-1998[f] 1999-2003[g]
yearling chinook salmon, wild	44.1	57.0	74.6	59.8	64.5	78.2	77.0	57.8	57.0	67.6	
subyearling chinook salmon, hatchery	-[h]	-	-	17.6	47.1	37.8	38.8	70.0	44.0	65.4	
subyearling chinook salmon, wild	29.2	8.4	34.2	18.8	45.3	50.6	63.4	67.0	48.1	62.8	
steelhead, hatchery	91.4	87.8	69.2	75.3	85.2	79.9	71.5	83.6	90.8	86.6	
steelhead, wild	70.5	52.4	50.1	50.9	55.3	50.5	42.6	65.0	72.9	73.8	
sockeye salmon, hatchery	-[d]	4.8	211	32.0	40.0	52.4	38.2	59.3	41.4	24.1	
sockeye salmon, wild	16.0	30.0	44.2	60.0	21.8	39.6	36.5	65.3	26.3	46.8	

[e] Hatchery sockeye not present prior to 1995.

[f] Baxter et al. 1999

[g] John Bailey, US Army Corps of Engineers-Walla Walla District, Pers. commun., November 2003

[h] Hatchery subyearling chinook salmon without PIT tags were not present until 1997.

Table 8. Mean separation efficiency values by comparison group (separator condition, light condition, and treatment) for juvenile salmonid length groups encountered during separation efficiency studies using conventional (McNary, McN) and test (insert) separators at McNary Dam, 2001. Values with the same letter superscript were significantly different (∝ = 0.05).

	Separator condition		Light condition		Treatment (separator condition, light condition)			
Length group	Insert	McNary	off	on	Insert, off	Insert, on	McN, off	McN, on
yearling chinook salmon								
<180 mm	60 (2.2)	58 (2.2)	56 (2.2)	61 (2.2)	56 (3.1)	64 (3.1)	56 (3.1)	59 (3.1)
≥180 mm	94 (2.1)[a]	69 (2.2)[a]	82 (2.0)	81 (2.2)	93 (2.8)	94 (2.8)	71 (2.8)	67 (2.8)
Total catch	64 (2.0)	59 (2.0)	60 (2.0)	64 (2.0)	61 (2.9)	68 (2.9)	59 (2.9)	60 (2.9)
steelhead								
<180 mm	65 (4..0)	63 (3.5)	62 (3.5)	67 (4.0)	63 (4.9)	66 (6.4)	59 (4.9)	67 (4.9)
≥180 mm	93 (1.4)[b]	78 (1.4)[b]	87 (1.4)	84 (1.4)	93 (2.0)	93 (2.0)	81 (2.0)	76 (2.0)
Total catch	91 (1.3)[c]	77 (1.3)[c]	85 (1.3	83 (1.3)	90 (1.8)	91 (1.8)	80 (1.8)	75 (1.8)
coho salmon								
<180 mm	32 (1.7)[d]	24 (1.7)[d]	28 (1.7)	28 (1.7)	32 (2.6)	32 (2.6)	24 (2.6)	24 (2.6)
sockeye salmon								
<180 mm	60 (3.5)[e]	35 (3.7)[e]	44 (3.7)	53 (3.5)	50 (5.3)	71 (5.3)	37 (5.3)	36 (5.3)
total yearling catch								
<180 mm	66 (1.9)	62 (1.9)	61 (1.9)[f]	67 (1.9)[f]	62 (2.7)[g]	69 (2.7)[g]	60 (2.9)[g]	64 (2.9)[g]
≥180 mm	93 (1.4)[h]	75 (1.4)[h]	85 (1.4)	83 (1.4)	93 (2.0)	93 (2.0)	77 (2.0)	72 (2.0)
Total catch	73 (1.2)[i]	69 (1.2)[i]	69 (1.2)[j]	73 (1.2)[j]	70 (1.7)	77 (1.7)	69 (1.7)	70 (1.7)
subyearling chinook salmon								
<180 mm	77 (3.6)	74 (3.6)	73 (3.6)	79 (3.6)	75 (3.5)	80 (3.5)	71 (3.5)	77 (3.5)

Since use of the insert did not result in biologically significant increased descaling or adverse stress levels (as measured by blood plasma indicators) compared to the McNary operational condition, it was installed for use in the Lower Monumental operational separator at the beginning of the 2002 outmigration. Over 2002 and 2003, using the insert has resulted in increased mean separation efficiency values for yearling chinook salmon, and sockeye salmon (Table 7).

Clarification of methods and theory underlying calculation of separation efficiency values is necessary to understand the metrics reported in Tables 7 and 8. The calculation used by smolt monitoring facilities (Table 7) is based on the assumption of species separation. For example, the upstream (A) separator section is classified as the 'chinook' section, and under 100% separation all chinook salmon are expected to be removed in that section. Similarly, all steelhead are expected to sound in the next (B) section. Therefore, smolt monitoring calculation of separation efficiency reflects that assumption:

$$ES_Y = Y_n / Y_t \times 100$$

Where:	ES_Y	=	Smolt monitoring separation efficiency estimate for species Y
	Y_n	=	Number of individuals in species Y exiting from the section assigned through which species Y is assumed to separate. Steelhead are assigned the downstream (B) section; all other species are assigned the upstream (A) section.
	Y_t	=	The total number of individuals of species Y exiting from both separator sections.

In fact, separation is a size dependent process, since all salmonid species seem to exhibit approximately similar behavioral propensity for sounding in a separator unit. Separation bars in the upstream section are set closer together so that only smaller fish can pass between them. An artifact of species composition in the smolt outmigration is that hatchery reared steelhead, representing the bulk of steelhead in the run, are generally larger than chinook, sockeye and coho salmon smolts and therefore relegated to sounding between the more widely spaced bars in the downstream section. However, most wild steelhead are sufficiently small to pass between the A section bars, and a substantial number of chinook and coho salmon smolts are too large to pass between the A section bars.

A more equitable separation efficiency calculation should therefore be based on fish size, rather than species. Under this scenario, all species are divided into two length groups (weight would also be a possible but less practicable metric to quantify). During separation efficiency studies, 180 mm fork length (FL) was selected as the most practical length criterion, since the majority of chinook salmon are smaller than 180 mm FL, while most steelhead are ≥180 mm FL. For example, of smolts sampled during 1998, 90% of yearling chinook salmon were <180 mm FL, and 86% of steelhead were ≥180 mm FL (Fig. 2). Using size class, separation efficiency should more appropriately be calculated based on the smolt size for which a given section was designed to separate.

$$ES_X = (AX_{<180} + BX_{\geq 180}) / X_t \times 100\%$$

Where:	ES_X	=	Separation efficiency estimate for species X.
	$AX_{<180}$	=	Number of individuals <180 mm FL in species X exiting from separator section A.
	$BX_{\geq 180}$	=	Number of individuals ≥180 mm FL in species X exiting from separator section B.
	X_t	=	Total number of individuals of species X captured from both separator sections.

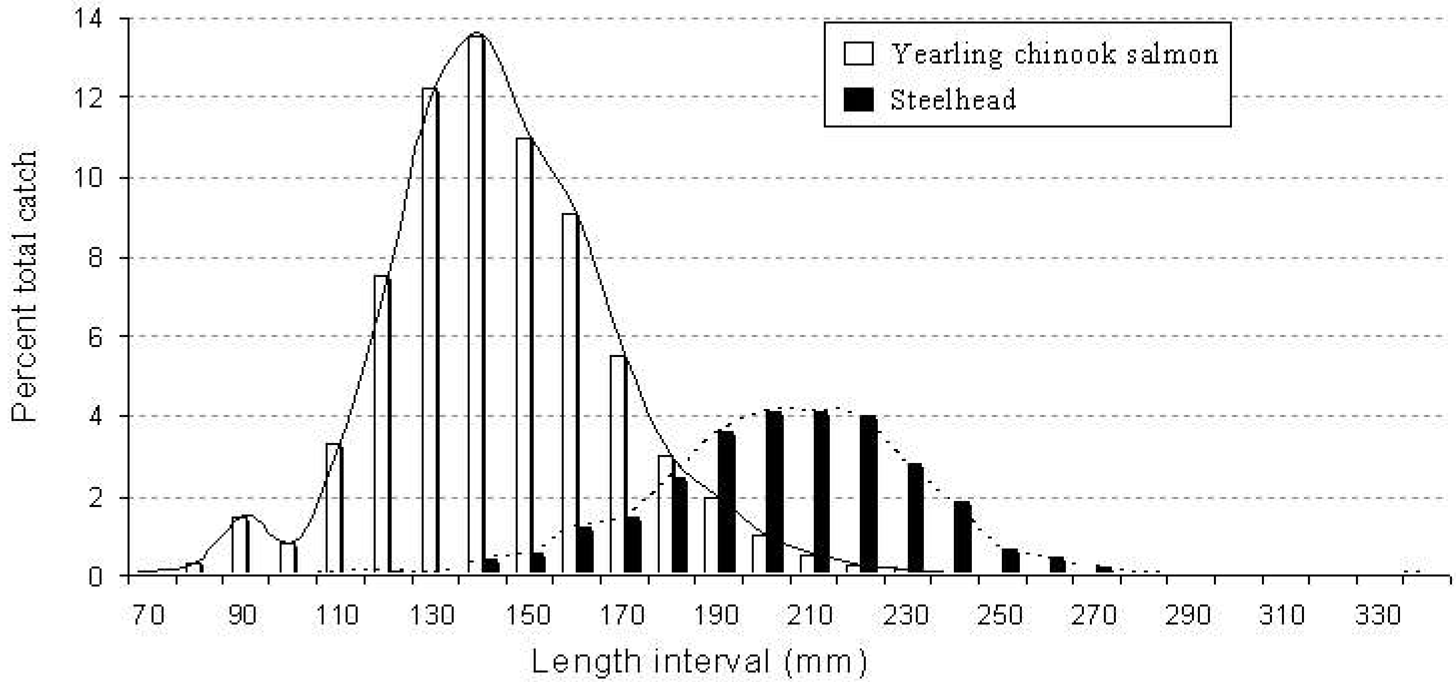

Figure 2. Length frequency distribution of yearling chinook salmon and steelhead sampled using McNary-style and high velocity flume evaluation separators during separation efficiency studies at McNary Dam, 1998.

In addition to studies aimed at improving the function and design of existing operational separators, a second approach explored alternatives to the existing separator design. The most promising alternative concept to emerge from interagency brainstorming sessions was the high velocity flume (HVF) design. Under this strategy, smolts enter a section of open flume directly after transport from the bypass channel. While traveling at velocities not normally present in current operational separator designs (1-2 m/s) , smaller smolts could sound between appropriately spaced separation bars within the flume, effecting separation from larger smolts unable to fit between the bars. Both groups would continue to different holding areas without the interruption caused by extreme velocity reduction, and without migration timing delays, stress, and fatigue induced by combating flows within the separator.

Results using an evaluation HVF separator at McNary Dam during the 1997 and 1998 juvenile salmonid outmigration period indicated that over 80% separation could be achieved for the total catch of all species combined at a transport velocity of 1 m/s, using separation bars submerged 50 mm below and parallel to the water surface, and spaced 19 mm apart (McComas et al. 2000, McComas et al. 2001). Based on these observations, a full scale prototype HVF separator was constructed at Ice Harbor Dam.

Results from cumulative evaluation of design and operational criteria using the prototype HVF separator from1999 through 2001 suggest that separation efficiency values of over 80% are possible for the total salmonid smolt catch, with moderate (5.3%) overall descaling, and virtually no timing delays in the separator unit (McComas et al. In prep b). Other studies have proposed a method for removing adults and large incidental fish prior to entering the smolt section of the HVF separator (McComas et al. In prep a). Though no high velocity flume separators are in operational use at juvenile bypass facilities in FCRPS dams, work is ongoing to complete the development of this concept as a possible option to the current separator design.

Diel Passage and Timing

Starting in the late 1960s, sampling in gatewells and at smolt monitoring facilities at powerhouses of Columbia and Snake River dams has consistently shown that the majority of smolts pass through the powerhouses at night (Long 1968, Sims and Ossiander 1981, Brege et al. 1996). The timing of this pattern can be affected by holding behavior in the gatewells and collection channels. Since then, hydroacoustic investigations (Johnson and Dauble 1995; BioSonics Inc. 1996, 1998; Ploskey et al. 1998) and radiotelemetry data (Vendetti and Kraut 1999) at many dams confirmed this was a consistent pattern, and provided more detail regarding specific hours of peak passage into turbine intakes. These studies have also shown that peak hours of passage change with different routes of passage. Passage at sluiceways at Bonneville and The Dalles Dams generally peaks in early morning hours.

Diel passage information for spillways is more cursory and not as consistent, paralleling powerhouse patterns at some dams, but showing high daytime passage and morning peaks at others. This subject is also discussed under the Spill Efficiency and Effectiveness section of Juvenile Passage Through Spillways.

Lower Granite Dam

Since 1994, both radiotelemetry and hydroacoustic studies have been conducted in the forebay of Lower Granite Dam to assess fish behavior and efficiency of a surface bypass collector (SBC) in front of turbine Units 4, 5, and 6. Johnson et al. (1998) found that fish passage into the SBC and spillway was higher during day hours than night hours. However, passage at the powerhouse pier nose and inside the turbine intake was reversed (Fig. 3). In 1999, spill was on from 1800 to 0600 and highest peak spillway passage occurred at 2300 hours. However, the diel distributions of fish passage were similar for the SBC and turbines, with peak passage occurring between 0600 and 1700 hours (Anglea 1999) (Fig. 4). Anglea et al. (1999) found that nighttime spill efficiency was 57%, and that downstream passage was slightly lower at night, while abundance of smolt-sized fish was higher during the day.

In 1998, a Behavioral Guidance Structure (BGS) was added to the SBC system in an attempt to guide more fish into the SBC. Anglea et al. (2000) reported that during the double/low configuration, multi-beam hydroacoustic monitoring upstream of the BGS entrance to the SBC revealed that fish moved in mostly a south direction during the day and night, with the exception of the near region closest to the entrance during daytime when they moved equally in both directions. During the double/high configuration, fish near the structure continued to move toward the south during both day and night as did the fish in the middle range at night. During daytime, slightly more fish moved north in the middle region. During daytime, fish in the far region moved equally in both directions.

Studies by Adams et al. (1997, 1999) have tracked radio-tagged spring chinook salmon as they approached the SBC. In 1998, 66 to 78% of all radio-tagged fish entered the fish bypass (via the turbine intakes) between 1900 and 0600 hours. With the exception of juvenile hatchery spring chinook salmon, most fish also entered the SBC

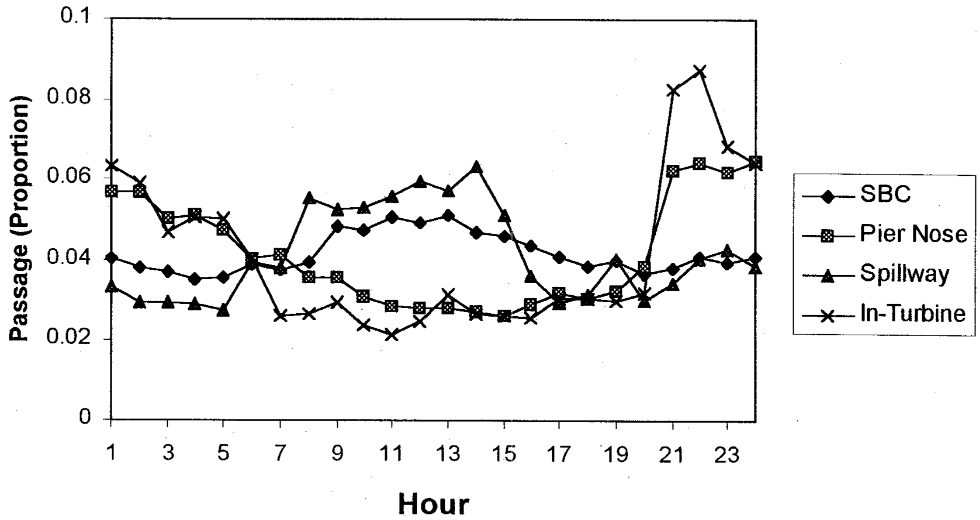

Figure 3. Diel distributions for surface bypass collector (SBC), pier nose, spillway, and in-turbine sample with fixed-location hydroacoustics at Lower Granite Dam, 1998 (from Johnson et al. 1998).

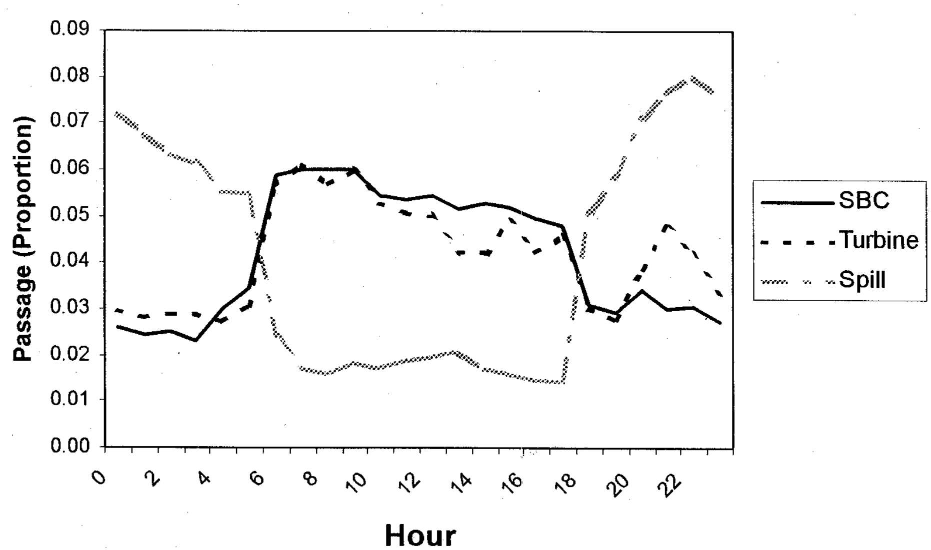

Figure 4. Diel distribution of fish passage for surface bypass collector, powerhouse, and spillway with fixed-location hydroacoustics at Lower Granite Dam, 1999 (from Anglea 1999).

during early evening and nighttime hours. Seventy-six percent of wild steelhead and wild fall chinook salmon, and 80% of hatchery steelhead entered the SBC between 1900 and 0600 hours.

Adams et al. (1997, 1999) also compared forebay residence times of hatchery chinook salmon, hatchery steelhead, and wild steelhead for the period 1994 to 1998. Flow and spill varied throughout the period, and were generally highest in 1997, and the peak spill in May was approximately 115 to 120 kcfs. Comparing forebay residence times for high flow years (1996 to 1998) to the lower flow years (1994 and 1995) suggests that in general, high flows and increased spill volume and hours reduced forebay residence time. For example, hatchery steelhead median residence times were 14.2 hours and 26.7 hours in 1995 and 1994, respectively, compared to 0.7 to 4.0 hours from 1996 to 1998. Hatchery yearling chinook median residence times were 6.8 hours steelhead median residence times showed little difference between years, and ranged from 0.7 to 3.5 hours between 1995 and 1998.

Angelea et al. (2002) provided a summary of all radiotelemetry and hydroacoustic studies conducted to evaluate the effectiveness of the SBC, including all information related to diel passage timing and movement.

Little Goose Dam

In 3 years of research by Vendetti and Kraut (1999), passage of radio-tagged juvenile fall chinook through the powerhouse at Little Goose Dam showed a slightly different diel pattern for passage (Fig. 5). Peaks in the number of detections were observed between 0500 and 0600 hours, and again in the evening between 1800 and 2000 hours, with the evening peak being the larger and longer of the two.

Lower Monumental Dam

To assess the downstream migrational patterns of smolts passing the dam and to estimate the effectiveness of spill in passing migrants, hydroacoustic investigations were conducted at Lower Monumental Dam from 1986 to 1989 (Wright et al. 1986, Ransom and McFaden 1987, McFaden 1988, Ransom and Sullivan 1989). As at other dams, turbine passage rates were highest during nighttime, with a peak at 2300 hours. During the 4 years of study, nighttime project passage (powerhouse and spill) ranged from 70 to 84%.

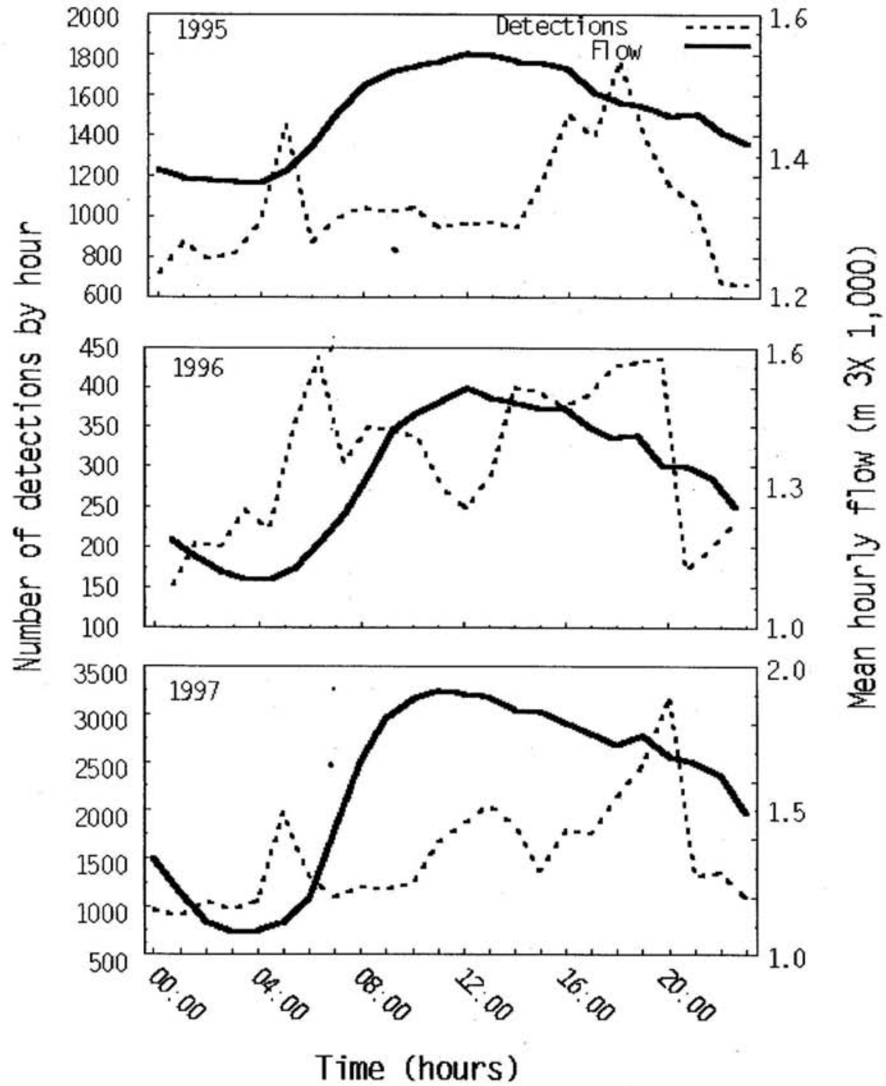

Figure 5. Total number of detections over the entire study period vs. mean hourly discharge at Little Goose Dam, 11 July-24 August,1995, 17 July-26 August,1996 and 15 July-23 August,1997 (from Venditti and Kraut 1999).

Spillway operation varied over the 4 years of tests. In 1986, spill varied from just nighttime spill (1800 to 0600 hours) to a constant 24-hour spill. Examination of diel passage during these different flow regimes suggested that both fish behavior and dam operations influenced the timing of spillway passage. During a period when spill was relatively constant throughout the day, a pronounced peak in passage occurred at 2200 hours. However, this evening peak was enhanced during periods when spill was off during daylight hours and started at 1800 hours. In studies from 1987 to 1989, the spillway was only operated from 1800 to 0600 hours and highest passage rates occurred at 2300 hours and declined steadily from 2300 to 0600 hours.

Ice Harbor Dam

Three years of hydroacoustic investigations at Ice Harbor Dam revealed diel passage patterns similar to other Columbia and Snake River dams (Johnson et al. 1983, Ransom and Ouellette 1988). Most migrants passed the dam at 2300 hours. Sluiceway diel passage rates were highest from 0600 to1300 hours. Turbine passage rates were highest from 2100 to 0600 hours.

During all 3 years of study, hourly passage rates through the spillway were more variable than through the turbine units or sluiceway. Generally, spillway passage rates were low in the early morning and then increased steadily to a peak at 1200 hours. The rate then declined rapidly, reaching a low point at 1700 hours, followed by a secondary peak at 2100 hours (only slightly lower than the 1200 hour peak).

In 1999, a radiotelemetry study of yearling chinook salmon passage was conducted at Ice Harbor Dam to determine tailrace egress and routes of passage under varying levels of spill and powerhouse flow (Eppard et al. in press). At 1800 h each day powerhouse flow was reduced while spill was increased to the maximum level based on dissolved gas levels, until 0600 h the following day. The test fish were released in the tailrace of Lower Monumental Dam each morning (0800 h). Receivers were positioned 1 km above Ice Harbor Dam and across the powerhouse and spillway, in the juvenile bypass channel, and on each submersible traveling screen. Each individual fish could be timed from the study entrance line through the passage route. Of the 580 fish detected 1 km above the dam, diel passage was fairly even distributed. A total of 302 and 278 fish passed the project during the day and night, respectively. Spilled fish had a lower forebay residence time, and fish first detected after dark had a lower forebay residence time than those first detected during daylight hours (Brad Eppard, NOAA Fisheries, Pers. commun., March 2000).

McNary Dam

No hydroacoustic or radiotelemetry studies have been conducted at McNary Dam that provide a robust source of diel passage information. Studies of orifice passage efficiency (OPE) using an orifice trap provide information on hourly passage from the gatewell(s) sampled. McComas et al. (1997) using an orifice trap found that passage from gatewells into the juvenile bypass channel appeared heaviest within a few hours of dawn and dusk, with the dusk peak having generally larger numbers of fish. Orifice trap data does not provide information on when fish first arrived in the forebay or entered the gatewell.

John Day Dam

As part of a smolt program, NOAA Fisheries has reported diel passage patterns of juvenile salmonids passing John Day Dam since the 1970s. This sampling, done mainly in gatewell Slots 3A and 3B, has shown that all species of juvenile salmonids tend to pass through the John Day powerhouse during hours of darkness (Brege et al. 1996). In more recent years, further observations by the Smolt Monitoring Program have confirmed predominant passage from 1800 to 0600 hours (Martinson et al. 1998) (Table 9).

Giorgi and Stevenson (1995) reviewed all gatewell sampling, hydroacoustic, and radiotelemetry data from 1980 to 1989. They concluded that even though none of the reports reviewed offered robust estimates of the diel passage, there was consistent agreement among all evaluation tools and across many years. The data indicated that smolts at John Day Dam exhibit a strong tendency to sound and pass through deep passage entrances during nighttime hours. Both the hydroacoustic and radiotelemetry data corroborated the gatewell sampling and also showed that timing of spillway passage through the deep spill intakes (47 to 58 ft below normal operating pool) paralleled powerhouse passage. One difference between the 1980s period that Giorgi and Stevenson reviewed and today is the amount of spill. In general, spill volumes have increased since implementation of the 1995 and 1998 Biological Opinions.

Another factor to consider is where the radio-tagged fish were released. For example, Stuehrenberg and Liscom (1983) released fish only a couple of miles upstream from the project. They found that fish released during the day for diel tests moved randomly downstream during the day with no obvious effect from rate of spill, although the sample size was extremely small (2). This pattern supported data based on diel sampling at the dam. When they combined 1982 data with data from 1980 and 1981, the effects of spill from 0 to 45% were observed. They saw an increase in the number of fish that passed the spillway when spill was increased. They concluded that spill should be greater than 34% of the total river flow to keep migrating smolts on the Washington side of the river.

In 1996, Holmberg et al. (1998) saw diel passage patterns similar to that reported by Giorgi and Stevenson (1995). However, the test fish were released only a short distance upstream (8 km) from the dam. In 1997, Hensleigh et al. (1999) saw a similar pattern for fish released a short distance upstream from John Day Dam. In contrast to previous years, they saw no apparent diel pattern for fish released at McNary Dam. They also saw that with high spill volumes in 1997, 50 to 75% of radio-tagged fish passed through the spillway, as compared to roughly 40% in previous years.

In 1997, BioSonics (1999a) saw higher numbers of fish passing the dam at night based on hydroacoustic studies. They estimated that 54 to 89% of the fish passed through the spillway. Also, during the spring when spill level was similar day and night, they observed that fish passed the spillway at a rather uniform rate. In 1999, Hansel and Beeman (1999) saw differences in species response to dam operations. Steelhead generally passed at night regardless of spill discharge. Further analysis of the data suggests that <200mm (primarily wild) steelhead did readily pass the spillway throughout the diel period (Gary Fredricks, NOAA Fisheries, Pers. commun., March 2000). Yearling chinook appeared to readily pass the project if daytime spill was provided. Information on John Day Dam spill effectiveness and efficiency is discussed above in the Spill Efficiency and Effectiveness section.

Time to pass the project is another important passage consideration. In 1995, Sheer et al. (1997) estimated that forebay residence time of spring chinook was 10 hours during conditions of low spill (< 14 kcfs). For 1996, Holmberg et al. (1998) reported delays of less than one hour during moderate spill conditions (47 to 125 kcfs). In 1997, Hensleigh et al. (1999) observed forebay residence times of 0.3 hours when spill discharge exceeded 300 kcfs.

To summarize diel passage information for John Day Dam, data collected during the 1980s suggested there was a strong and consistent tendency for fish to pass the project at night. However, it is difficult to compare results from recent and earlier studies, due to differences in sample sizes, release points for tagged fish, and radio-tagged fish release timing. Based on radiotelemetry and hydroacoustic study results since 1997, it appears that yearling migrants, especially yearling chinook, will readily pass the spillway during the day if spill volumes are 30% or greater. Daytime spill volumes other than 30% have not been tested using recent release protocols for radio-tagged fish.

More recently, Hansel and Beeman (1999) found that juvenile steelhead generally passed at night regardless of the spill discharge at their arrival. Yearling chinook salmon passed similarly when arriving during daytime spill discharge of 68%, but during 30% daytime spill they passed at that time. Fish arriving at night generally passed at night.

Radiotelemetry studies conducted in 1999 and 2000 were the only studies that collected enough data to discern diel patterns of passage (Anglea et al. 2001). This was because there were high enough numbers of tagged fish entering the forebay throughout the 24-h blocks studied to depict diel passage patterns. Because a majority of juveniles passed the dam at night (especially steelhead where often over 90% passed at night regardless of treatment), it is important to examine and compare day and night differences. They show that the addition of day spill certainly increases the proportion of juvenile salmonids passing the spillway during the day (especially yearling chinook salmon in 2000).

Table 9. Percent night passage (1800-0600 hours) for each season at John Day Dam, 1985 to 1997 (Martinson et al. 1998).

Year	Yearling chinook	Subyearling chinook	Wild steelhead	Hatchery steelhead	Coho	Sockeye
1985	83.2	83.7	N/A	N/A	91.0	86.8
1986	75.5	80.1	N/A	N/A	95.9	81.9
1987	84.5	85.4	93.6	85.6	95.0	94.9
1988	80.0	80.7	80.8	70.3	83.9	87.1
1989	86.4	86.2	73.6	79.4	93.0	79.0
1990	79.7	84.4	76.3	94.8	95.6	85.0
1991	89.9	77.0	91.0	92.3	96.2	83.6
1992	82.8	78.7	95.3	91.5	96.0	94.9
1993	83.3	87.8	83.4	80.7	95.1	86.5
1994	80.9	68.1	91.6	81.4	92.2	94.5
1995	80.7	79.7	87.9	75.8	91.5	79.5
1996	68.6	70.0	81.6	74.7	80.2	65.6
1997	62.6	73.1	67.0	70.6	73.7	59.6
AVERAGE	79.8	79.6	83.8	81.5	90.7	83.0
MIN	62.6	68.1	67.0	70.3	73.7	59.6
MAX	89.9	87.8	95.3	94.8	96.2	94.9

Unfortunately, nearly the same proportion of fish that passed the spillway during the day would have been guided to the juvenile bypass system if they passed at night resulting in no significant change in FPE. There may still be a benefit to day spill, though, because of the considerable reduction in passage delay of juvenile chinook salmon arriving during this time.

The Dalles Dam

Diel passage through the powerhouse at The Dalles Dam is similar to the pattern seen at other Columbia and Snake River dams. Both gatewell dipping (Long 1968) and hydroacoustics (Magne et al. 1983, Steig and Johnson 1986, Johnson et al. 1987, BioSonics Inc. 1996) documented that the majority of all salmonids entered gatewells between 1900 and 0700 hours. In studies conducted by BioSonics Inc. (1996), both spring and summer migrants exhibited peak passage during hours of darkness (Figs. 6-7).

As with Bonneville Dam, diel passage through the sluiceway does not seem to follow the nighttime passage patterns. Early studies by Nichols (1979) and Nichols and Ransom (1980, 1981) reported that sluiceway passage peaked during daylight hours, typically around mid-day. More recent hydroacoustic studies have also found that average daylight passage through the sluiceway was the predominant pattern at The Dalles in the spring (Fig. 6). During the summer period, passage rates through the sluiceway also showed low night passage, but the peak was during the afternoon rather than morning (Fig. 7). After reviewing all available literature, Giorgi and Stevenson (1995) concluded that regardless of the sampling approach employed, during the spring, juvenile salmonids pass via the sluiceway at The Dalles Dam through most of the 24 hour period with peak passage during daylight hours.

There are few measures of diel passage at The Dalles Dam spillway, because in most of the studies conducted there, spill was only provided at night and did not span a 24-hour period. However, in studies conducted in 1996 by BioSonics Inc. (1996), spill levels were maintained for 24-hour periods and slightly higher morning passage rates via spill were found during the spring (Fig. 6). This morning peak was even more pronounced during the summer migration (Fig. 7). In 1999, Ploskey et al. (1999) found that diel distribution of fish passage during the spring through the spillway was similar between the 30 and 64% spill treatments. The proportion of spillway fish passage was relatively uniform except for a substantial peak between 2000 and 2100 hours. During the summer fish passage through the spillway was again fairly uniform except for peaks at 0600-0700 and 2000-2200 hours during 30% spill, 0200 and from 1800-1900 hours during 64% spill.

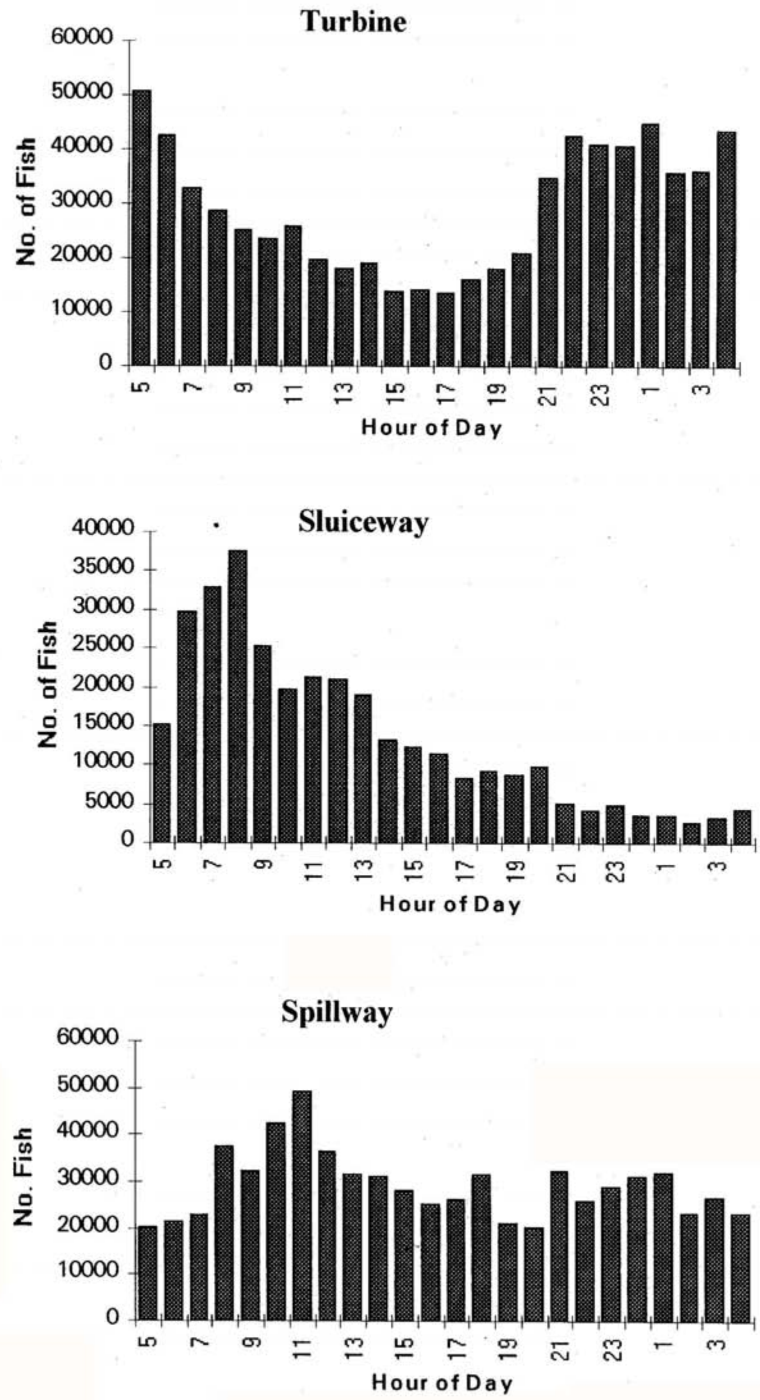

Figure 6. Diel variation in passage through turbines, sluiceway, and spill at The Dalles Dam, spring 1996 (from Biosonics 1996).

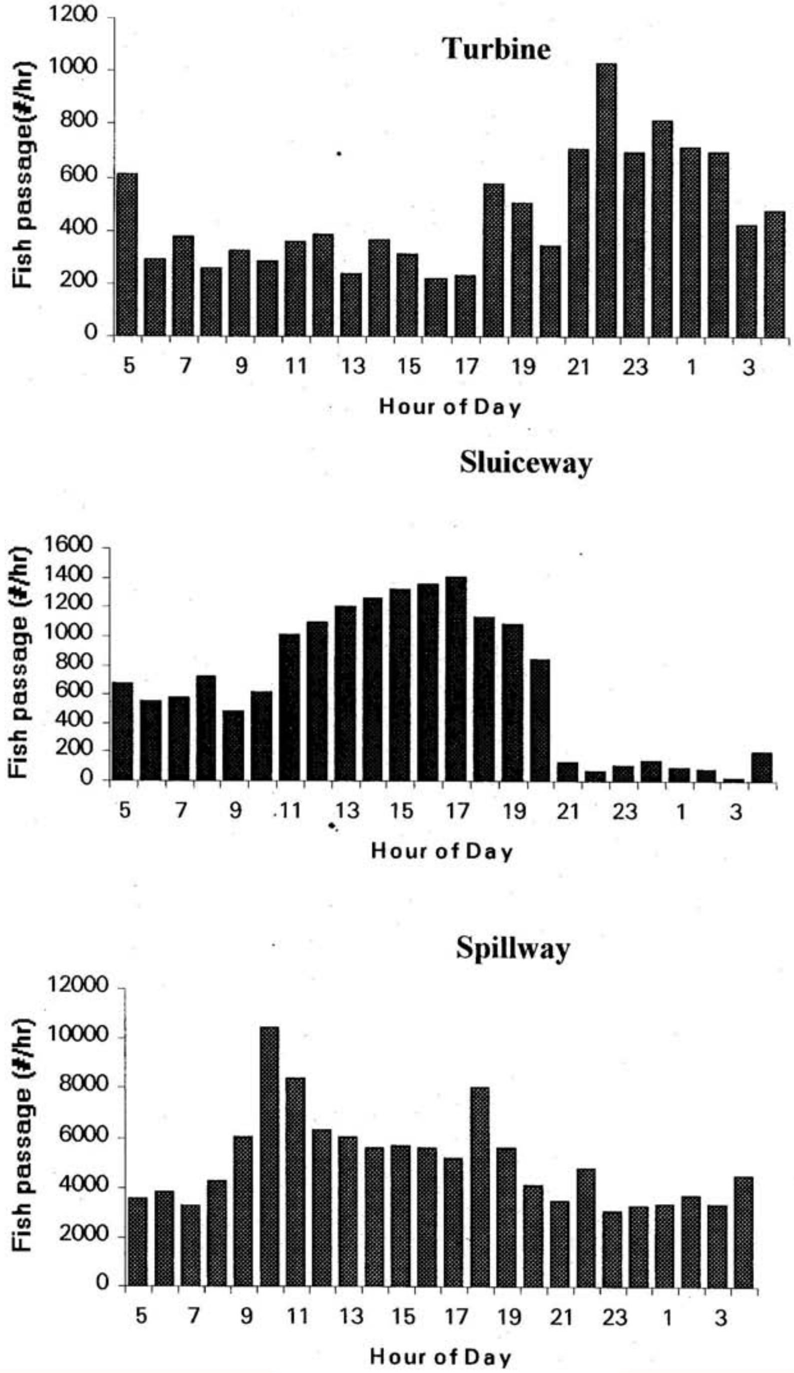

Figure 7. Diel variation in passage through turbines, sluiceway, and spill at The Dalles Dam, summer 1996 (from Biosonics 1996).

More recently, Hansel and Beaman (1999) examined FPEs and spill and sluiceway passage efficiencies of yearling chinook salmon and steelhead at the Dalles Dam. With the exception that the FPE of juvenile steelhead was significantly greater during the day than night during the 30% spill discharge (95 vs 86%, respectively), neither FPE nor spill passage efficiency differed significantly between day and night for either species. However, differences between diel passage conditions did affect spillway passage location. During the adult spill pattern, radio-tagged fish passed mostly through the southern end of the spillway, whereas during the juvenile spill pattern they passed predominantly through the northern end. Median residence time in the near-dam forebay area was generally less than 0.4 h for both species, but juvenile steelhead during the daytime 30% spill discharge had median residence time of 2.3 hours.

With hydroacoustics, Moursand et al. (2001) used relative passage rates between the major routes of turbine, sluice, and spillway to examine temporal differences in passage. Turbine passage peaked at night in the spring, but that peak shifted to afternoon by summer (and). Spillway passage showed a peak in the morning hours, and this trend remained through the summer. There was a pronounced trend for fish to preferentially pass through the sluice during the day rather than at night. This trend was clear in both spring and summer.

Bonneville Dam

Passage patterns at Bonneville Dam are more complex than at other dams, because there are three separate structures, two powerhouses and an unattached spillway. There are also two passage routes at each powerhouse, the turbines and ice/trash sluiceway. For this reason, the available information is provided separately for each powerhouse and the spillway.

First Powerhouse–Gessel et al. (1986) dipped gatewells over 24 hours during six periods throughout the spring migration and observed peaks in passage shortly after dawn and dusk, with the evening peak being typically much higher. From 1992 to 1995, the Smolt Monitoring Program sampled juvenile migrants moving through the downstream migrant channel on a 24-hour basis (Martinson et al. 1999). During this time, passage for all species increased at dusk, at about 2000 hours, and peaked at 2200 hours (Table 7). Since 1995, the Smolt Monitoring Program has sampled for 8 hours per day only (from 1600 to 2400 hours) and has reported the typical dusk peak in turbine passage observed at Snake River and lower Columbia River dams (Martinson et al. 1996, 1997, 1998).

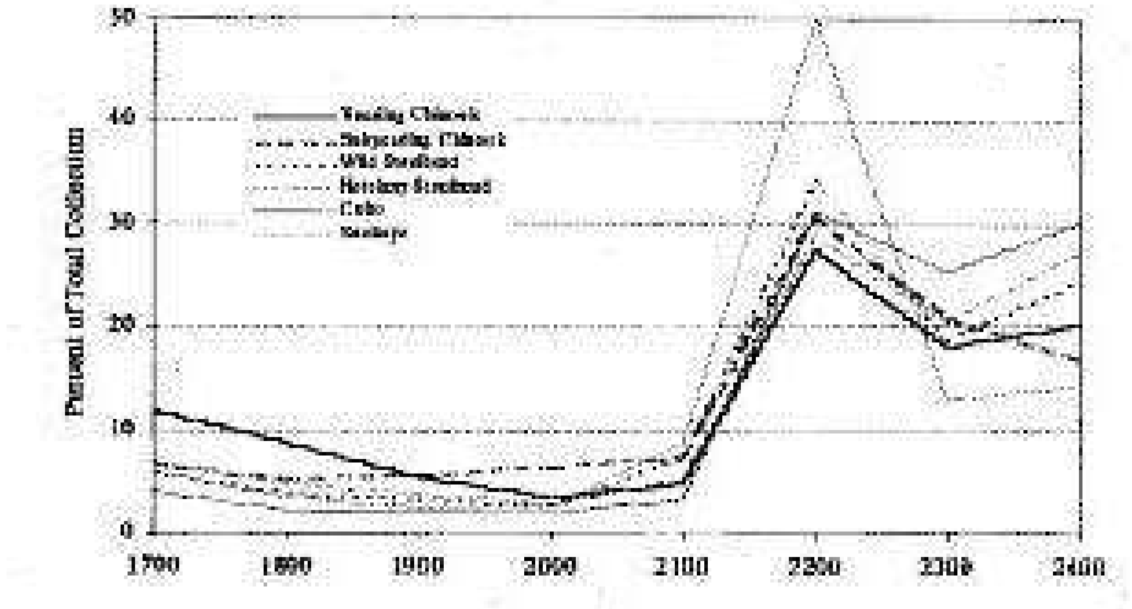

Figure 8. Eight-hour passage patterns for all species at Bonneville Dam First Powerhouse, 1998 (from Martinson et al. 1998).

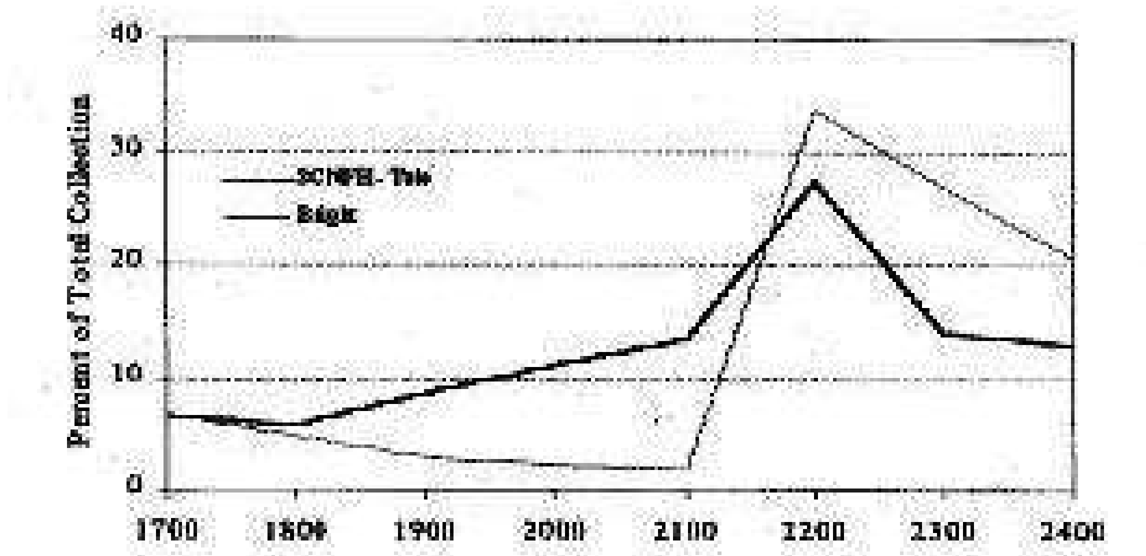

Figure 9. Eight-hour passage patterns of subyearling chinook stocks at Bonneville Dam First Powerhouse, 1998 (from Martinson et al. 1998).

Table 7. Percent night passage (1800-0600 hours) for 1992 to 1995 at Bonneville Dam First Powerhouse (Martinson et al. 1999).

Year	Yearling chinook	Subyearling chinook	Wild steelhead	Hatchery steelhead	Coho	Sockeye
1995	52.8	65.8	68.3	62.9	70.7	56.2
1994	49.6	52.4	52.2	53.1	66.7	74.6
1993	43.2	56.2	67.1	62.4	68.1	63.6
1992	52.0	44.0	62.3	61.3	60.0	69.0
MEDIAN	50.8	54.3	64.7	61.9	67.4	66.3
MIN	43.2	44.0	52.2	53.1	60.0	56.2
MAX	52.8	65.8	68.3	62.9	70.7	74.6

In 1998, Martinson et al. (1999) observed that the 8-hour passage patterns of tule and bright stocks of subyearling chinook salmon followed the same general pattern as spring migrants with only slight differences (Figs. 7 and 8). Increases in tule passage began earlier, at about 1800 hours, had a smaller peak at 2200 hours, then averaged a lower percent of total passage for the remaining hours. Upriver bright passage resembled the spring migrants, with a larger, more abrupt increase at 2200 hours and a gradual decline in passage thereafter.

Hydroacoustic data from Ploskey et al. (1998) also showed that mean hourly smolt passage into turbines at the First Powerhouse was higher during night hours than during day hours in 1996 and 1997. The data were more variable in spring than in summer. Ploskey et al. (1998) also observed that diel passage through the sluiceway at the First Powerhouse varied from turbine passage. During both the spring and summer smolt migrations, the majority of passage through the sluiceway occurred during the early morning hours with a peak at approximately 0300 hours. Passage was reduced during daytime hours, with a secondary peak shortly after sunset.

Second Powerhouse--The patterns of diel variation in fish passage rates during the spring migration, seen at the First Powerhouse, do not seem to be as strong or consistent at the Second Powerhouse. Studies by BioSonics Inc.(1998) showed little diel variation at Unit 11A (both guided and nonguided) with no peak at dusk. Spring studies by Ploskey et al. (1998) showed a nighttime peak for a couple of treatment days, but the majority of days had highly variable data, with no consistent pattern.

During summer migration studies by BioSonics Inc. (1998), subyearling chinook salmon exhibited a minor diel variation in Unit 11A with highest fish passage between 2100 and 2200 hours. However, Ploskey et al. (1998) observed a pronounced nighttime peak from 2200 to 2300 during summer smolt passage through the turbines at the Second Powerhouse.

Similar to the First Powerhouse sluiceway, spring time fish passage through the sluice chute at the Second Powerhouse occurred during early daytime, between 0600 and 1300 hours (BioSonics Inc. 1998). A second smaller increase in passage rates was observed between 1800 and 2100 hours. During the summer migration, the peak passage rates through the sluice chute occurred between 0600 and 1800 hours. The hours of lowest passage through the sluice chute were from 2100 to 2200 hours, coinciding with the hours of highest fish passage of guided fish into Unit 11A.

Spillway--A limited amount of hydroacoustic data is available pertaining to diel timing of juvenile salmonids through the spillway. BioSonics Inc. (1998) found that fish passage rates were higher during nighttime at the spillway in the spring. These results were not adjusted for changes in spill levels; however, because of high river flows, spillway discharges showed little variation during the spring. During the summer, fish passage rates were also highest at night at the spillway; however, this was in part due to spillway operations, when Spillbay 5 (with transducer) was typically closed between 0500 to 2200 hours.

In 1999, Plumb et al. (2001) found that regardless of release site more fish were first detected at the spillway (37%) than at the First Powerhouse (31%) or Second Powerhouse (29%). Median residence times for hatchery steelhead were 5.6, 3.9, and 0.3 hours for the two Powerhouses and the spillway, respectively. Corresponding times for yearling chinook salmon were 1.0, 0.2, and less than 0.1 hours. Additionally, fish from all releases passed the dam throughout the diel period, with slightly higher numbers during night passage (2000-0659 hours).

Evans et al. (2001a) found that passage rates were highest for both species during the day at the spillway and at the First Powerhouse, but nighttime passage was the highest at the Second Powerhouse. Also, the median residence times in the forebays of Bonneville Dam ranged from 8 minutes to 9.7 hours depending on species and forebay area. Discharge rates as well as diel periods effected residence times for both species. In a more recent study, Evans et al. (2001b) found that median forebay residence time was shortest for the Second Powerhouse (0.7 hours) compared to 2.4 hours at the First powerhouse and 1.5 hours at the spillway. Again, a higher proportion of all groups passed during the night.

Water Temperature Effects

The state water quality standard for water temperatures on the Snake and Columbia Rivers calls for temperature to not exceed 68°F. The upper incipient lethal water temperature for salmonids is defined as 77°F (Coutant 1999). Water temperature data are collected daily at mainstem Columbia and Snake River dams where Smolt Monitoring Program activities occur, from April to October at lower Snake River dams and into December at McNary Dam. Temperatures are taken at the juvenile fish bypass system sampling facility at 0700 hours, which corresponds to the beginning/end of the daily sample period. Since the data are collected early in the morning, reported temperatures do not represent the diurnal temperature cycle, and likely underestimate daily maximums. In addition to the daily data collection at the juvenile fish bypass sampling facility, at McNary Dam during July and August, detailed water temperature data are collected by thermistors by on-site fishery agency biologists under contract to COE. At the powerhouse, thermistors record data from northern, middle, and southern locations in the juvenile fish collection gallery and turbine intake gatewells. At the juvenile fish bypass facility, thermistors record temperature data from a representative raceway, the wet separator, and the barge-loading dock. These data show that horizontal, vertical, and diurnal temperature gradients occur between these sampling locations.

Water temperature issues are most prominent and of concern during July and August when ambient air temperatures can exceed 90ºF. Acute fish mortality problems have been observed during times of high water temperatures. In 1994, the water temperature at the McNary Dam juvenile fish facility reached 68ºF in mid-July. A few days later, the facility experienced a massive fish mortality, most likely related to river water temperatures exceeding 70ºF. Prior to 16 July 1994, thermal profile data showed elevated water temperatures but not the large thermal gradients between operating and non-operating turbine units associated with fish mortality observed in previous years (Hurson et al. 1996).

The NOAA Fisheries 1995 BIOP required the action agencies to take measures to reduce the potential for reoccurrence of the 1994 incident (NMFS 1995a). At McNary Dam from 10 to 12 July 1998, daily facility mortality was 3 to 8% (34,900 of 831,700 fish collected), which was coincident with water temperatures above 70^0 F. Also in 1994, daily average river temperatures in the lower Columbia River migration corridor reached 74ºF during late-July. Coutant (1999) suggested that the cause of these acute mortalities observed at McNary Dam was a cumulative thermal dose of exposure to high-temperature water, received over a period of several days.

Elevated water temperatures likely contribute to juvenile fish facility loss during summer months. In July 1998, facility mortality of wild subyearling fall chinook salmon collected at lower Snake River collector dams ranged from 1.5 to 3.8% for Lower Monumental and Little Goose Dams, respectively. In 1997, it ranged from 2.1 to 7.7% for Lower Monumental and Little Goose Dams, respectively. High water-temperatures and outbreaks of disease later in the season likely contribute to increased facility mortality. Total juvenile fish facility mortality during the summer at McNary Dam was 2.2% during 1997 and 1998 (Hurson et al. 1999).

To address these temperature issues, a regional Water Quality Team has been formed, which is co-chaired by the Environmental Protection Agency and NOAA

Fisheries. The Team is working on a plan to address alternative operations during warm water periods at McNary Dam, with the goal of developing an approved plan by 2000.

In addition to conditions at juvenile bypass facilities that occur within periods of elevated water temperature during the summer months, recent overall trends in river temperatures have been evaluated. ODFW and WDFW (1998) noted an increasing trend in the daily average water temperature for Bonneville Dam during August and September from 1938 to 1997. While it is possible this increase is a result of the development of the hydropower system, the trend is confounded by the construction and operation of both Federal and non-Federal storage facilities, climatic changes, and possibly increased irrigation withdrawals during the period. The trend could result from one or many of these factors.

Effects of Bypass Systems on Smolt Condition

During the 1990s, two existing juvenile fish collection and bypass facilities at dams on the Snake and Columbia Rivers were replaced with new facilities (Little Goose Dam (1990) and McNary Dam (1994)), and new juvenile fish collection and bypass facilities were built at Lower Monumental Dam (1993), Ice Harbor Dam (1996), John Day Dam (1998), and Bonneville Second Powerhouse (1999). Each of these new facilities were tested to locate and correct any areas within the structures, flumes, and pipes that might cause descaling, injuries, or mortalities to juvenile fish passing through the facilities. In addition, the region's state and federal fishery agencies monitor daily samples of the fish for descaling, injury, and mortality to ensure that these facilities are operated in a safe manner.

Because the 1993 smolt outmigration through the Snake River was the first where all hatchery spring/summer chinook salmon were adipose-fin clipped, that year is used as a starting point for looking at fish condition (descaling and mortality) at each of the Snake River dams. Fish condition at the Snake River dams and at McNary Dam is reported in Hurson et al. (1994) and Hurson et al. (1998). More recent fish condition information is collected as part of the annual juvenile fish transportation program at each of the Snake River dams and at McNary Dam and is summarized below (David Hurson, Pers. commun., U.S. Army Corps of Engineers).

The Lower Granite Dam juvenile fish facility was not replaced during the 1990s. Descaling rates from 1998 to 2002 averaged 2.7% for hatchery spring/summer chinook salmon, **1.5**% for wild spring/summer chinook salmon, 2.8% for subyearling chinook salmon, 4.1% for hatchery steelhead, 1.7% for wild steelhead, 5.7% for sockeye salmon, and 3.0% for coho salmon (declared extinct in the Snake River in the early 1980s, but coho salmon were re-introduced for the 1996 outmigration). Mortality rates from 1998 to 2002 averaged 0.3% for hatchery spring/summer chinook salmon, 0.3% for wild spring/summer chinook salmon, 1.2% for subyearling chinook salmon, less than 0.1% for hatchery steelhead, less than 0.1% for wild steelhead, 1.6% for sockeye salmon, and 0.3% for coho salmon.

Descaling rates at Little Goose Dam from 1998 to 2002 averaged 6.2% for hatchery spring/summer chinook salmon, 4.9% for wild spring/summer chinook salmon, 3.2% for subyearling chinook salmon, 6.7% for hatchery steelhead, 3.5% for wild steelhead, 10.1% for sockeye salmon, and 6.4% for coho salmon. Mortality rates from 1998 to 2002 averaged 0.6% for hatchery spring/summer chinook salmon, 0.7% for wild spring/summer chinook salmon, 6.1% for subyearling chinook salmon,0.3% for hatchery steelhead, 0.2% for wild steelhead, 2.0% for sockeye salmon, and 0.7% for coho salmon.

Descaling rates at Lower Monumental Dam from 1998 to 2002 averaged 3.1% for hatchery spring/summer chinook salmon, 2.5% for wild spring/summer chinook salmon, 2.3% for subyearling chinook salmon, 4.5% for hatchery steelhead, 2.7% for wild steelhead, 3.7% for sockeye salmon, and 3.9% for coho salmon. Mortality rates from 1998 to 2002 averaged 0.2 % for hatchery spring/summer chinook salmon, 0.2% for wild spring/summer chinook salmon, 2.3% for subyearling chinook salmon, 0.3% for hatchery steelhead, 0.2% for wild steelhead, 0.7% for sockeye salmon, and 0.1% for coho salmon.

The juvenile bypass system for Ice Harbor Dam was completed and evaluated in 1996. Hatchery steelhead and yearling chinook salmon released at two locations in the collection channel were recaptured and evaluated. Mortality and descaling for both species were less than 0.2% (Gessel et al. 1997). This facility is currently only operated for short periods during parts of each outmigration each year. While this provides for a cursory examination of descaling and mortality annually, the small numbers of fish examined minimize the usefulness of any data collected.

The new juvenile fish facility at McNary Dam, the first Columbia River dam encountered by fish exiting the Snake River, became operational in 1994. The hatcheries in the Columbia River do not adipose-fin clip all of their spring/summer chinook salmon; therefore, the juvenile fish facilities at Columbia River dams cannot distinguish between hatchery and wild fish. Descaling rates at McNary Dam from 1998 to 2002 averaged 6.4% for spring/summer chinook salmon, 2.9% for subyearling chinook salmon, 5.7% for hatchery steelhead, 3.7% for wild steelhead, 9.6% for sockeye salmon, and 3.9% for coho salmon. Mortality rates from 1998 to 2002 averaged 0.2% for spring/summer chinook salmon, 1.0% for subyearling chinook salmon, 0.4% for hatchery steelhead, 0.3% for wild steelhead, 0.4% for sockeye salmon, and 0.4% for coho salmon.

Measures of bypass system survival at John Day Dam were comprised of observations of mortality at the sampling facility (Martinson et al. 2003) and data gathered during a bypass system evaluation (Absolon et al. 2000a.). Descaling rates at John Day Dam from 1998 to 2002 averaged 3.9% for yearling chinook salmon, 1.1% for subyearling chinook salmon, 1.9% for wild steelhead, 6.1% for hatchery steelhead, 2.5% for coho salmon, and 8.2% for sockeye salmon. Mortality rates from 1998 to 2002 through the modified bypass system reported by Martinson et al. (2003) at the John Day Dam sampling facility averaged 0.5% for yearling chinook salmon, 0.2% for subyearling chinook salmon, 0.1% for wild steelhead, 0.3% for hatchery steelhead, 0.2% for coho salmon, and 0.8% for sockeye salmon.

The latest modifications to the juvenile bypass system at John Day Dam were completed in April 1998. Post-construction evaluation of the system was conducted by NOAA Fisheries in 1998 and 1999 (Absolon et al. 2000a, 2000b.). As part of this evaluation in 1998, hatchery chinook salmon yearlings and steelhead were released at various points within the system. Direct mortality during passage from the collection channel to the evaluation facility ranged from 0 to 1.5% for yearling chinook salmon. Direct mortality for three collection channel releases of steelhead ranged from 0 to 1.5%. In 1999, three groups of chinook salmon fry were released into the upstream end of the elevated flume and recaptured in the sample tank. No descaling or injuries were observed on any of the recaptured fish.

At Bonneville Dam, mortalities at the First- and Second-Powerhouse sampling facilities were noted by Martinson et al. (2003). Descaling at the First Powerhouse from 1988 to 2002 averaged 5.0% for yearling chinook salmon, 1.7% for subyearling chinook salmon, 4.2% for wild steelhead, 9.6% for hatchery steelhead, 3.6% for coho salmon, and 20.3% for sockeye salmon. First-Powerhouse mortalities from 1988 to 2002 averaged 0.2% for yearling chinook salmon, 0.4% for subyearling chinook salmon, 0.1% for wild steelhead, 0.1% for hatchery steelhead, 0.1% for coho salmon, and 0.5% for sockeye salmon. The latest modifications to the juvenile bypass system at the Second Powerhouse became operational in 2000. Descaling observed at the Second Powerhouse from 2000 to 2002 averaged 2.5% for yearling chinook salmon, 0.5% for subyearling chinook salmon, 1.9% for wild steelhead, 5.5% for hatchery steelhead, 1.0% for coho salmon, and 6.6% for sockeye salmon. Mortalities observed at the Second Powerhouse from 2000 to 2002 averaged 0.8% for yearling chinook salmon, 0.5% for subyearling chinook salmon, 0.3%

for wild steelhead, 0.4% for hatchery steelhead, 0.9% for coho salmon, and 2.0% for sockeye salmon. Although these estimates give some measure of the upper limit of direct, immediate bypass system passage mortality, they cannot reflect mortality through the entire system since sampling locations are typically some distance upstream from the outfall. Also, some mortalities observed within a system may have resulted from prior injuries and conversely, some live fish observed in samples may die of passage effects at a later time.

Passage survival studies conducted by NOAA Fisheries at Bonneville Dam from 1987 through 1990 and in 1992 involved releases of differentially marked subyearling chinook salmon into various passage routes at Bonneville Dam, including the Second Powerhouse bypass system (1987 to 1990) and the First Powerhouse bypass system (1992). These studies used fish identified with freeze brands and marked with coded-wire tags. Relative short-term survival for treatment groups was based on data from recapture of juvenile test fish at Jones Beach (RKm 74) during seaward migration. Relative long-term survival for test fish using the various passage routes was to have been based on recoveries of coded-wire tags from fisheries, hatcheries, and surveys of spawning areas.

Results from the NOAA Fisheries relative survival studies at Bonneville Dam were reported in Dawley et al. (1988, 1989), Ledgerwood et al. (1990, 1991, 1994), and Gilbreath et al. (1993). Results from Second Powerhouse tests conducted by NOAA Fisheries from 1987 to 1990 indicated that relative survival of bypass-released groups averaged 8.3% less than reference groups released in the immediate tailrace downstream from the dam, 7.6% less than turbine releases, and 17.3% less than groups released approximately 2 miles downstream from the dam (Dawley et al. 1996). Using a similar methodology, NOAA Fisheries released subyearling chinook salmon at the Bonneville Dam first powerhouse in 1992 (Ledgerwood et al. 1994). Results indicated that relative survival of bypass-released groups was 11.8% less than turbine-released groups and 28.3% less than groups released approximately 2 miles downstream from the dam.

Due to the unforseen low relative survival of bypass-released groups, NOAA Fisheries conducted a separate evaluation at the Bonneville Second Powerhouse bypass system from 1990 to 1992 (Dawley et al. 1998a). In these studies, referred to as direct assessments of passage survival, a trap-net attached to the submerged outfall was used to capture fish immediately upon exit from the bypass system. Results indicated that the bypass system was not responsible for the considerable survival differences noted in the previous paragraph, although descaling, injury, and stress among release groups did increase as the distance traveled through the bypass system increased. It was hypothesized that increased stress and fatigue of fish traveling through the bypass system, in combination with a poor outfall location, resulted in high predation rates on juvenile salmonids exiting the bypass system.

Based on the results of these studies, the COE modified the Bonneville Second Powerhouse bypass system in 1999. The collection channel was modified to improve hydraulic conditions, a new dewatering facility was installed which uses floor and wall screens, and a new 1.7-mile long conveyance pipe and outfalls were constructed at Hamilton Island downstream of Bonneville Dam. Gilbreath and Prentice (1999) evaluated the new bypass system. They found that none of the 305 hatchery-reared steelhead were dead, injured, or descaled. A total of 332 run-of-the-river steelhead showed no signs of passage effects other than one mortality. Median passage time from the upstream end of the collection channel to Hamilton Island was about 3 hours for hatchery-reared steelhead and about 1 hour for run-of-the-river steelhead. The median passage time through the length of the collection channel was about 1 hour for hatchery-reared steelhead, as compared to only a few minutes for run-of-the-river steelhead. For yearling chinook salmon, they found that of the 394 hatchery-reared chinook salmon released, none were injured or descaled. However, 5 were dead which they attributed to tagging and handling. A total of 209 run-of-the-river yearling chinook salmon were examined, of which 5 were descaled and one was dead after passage through the new system.

Median passage time from the upstream end of the collection channel to Hamilton Island was about 46 minutes for both hatchery-reared and run-of-the-river chinook salmon. The median passage time through the length of the collection channel was about 4 to 7 minutes for hatchery-reared and run-of-the-river chinook salmon, respectively. During the summer outmigration period 351 run-of-the-river subyearling chinook salmon were recaptured. None were descaled or injured, and 3 fish were recovered dead. Median passage time from the upstream end of the collection channel to Hamilton Island was about 42 minutes for run-of-the-river subyearling chinook salmon. The median passage time through the length of the collection channel was 5 minutes for run-of-the-river subyearling chinook salmon.

Effects of Bypass Systems on Blood Chemistry

Stress, generated by external and internal stimuli, induces quantifiable biochemical responses in fish (Hane et al. 1966, Grant and Mehrle 1973). Clinical evaluation of blood plasma has associated stress with changes in concentrations of cortisol and adrenaline, which influence levels of secondary indicators including lactate, glucose, liver glycogen, leucocyte count, free fatty acids, and the balance of various electrolytes (Mazeaud et al. 1977).

Several stressors related to passage through fish bypass facilities at hydroelectric dams have been shown to alter indicator concentrations in juvenile salmonids under experimental conditions. For example, elevated plasma cortisol and glucose levels have been associated with crowding and handling (Wedemeyer 1976, Congleton et al. 1984), descaling (Gadomski et al. 1994), acclimation temperature (Barton and Schreck 1987a), and confinement (Strange et al. 1978).

Under field conditions, Maule et al. (1988) demonstrated that juvenile chinook salmon plasma cortisol and glucose concentrations could increase cumulatively as the fish passed successive points in the bypass system, corroborating similar results from laboratory work with sequential handling (Barton et al. 1986, Congleton et al. 1999). However, Congleton et al. (1999) found that, generally, peak plasma cortisol level in fish following handling experiments did not appear to be affected by repeated stressful experiences. They point out it is possible that fish in the multiple stress groups experienced some form of chronic, low-level stress, that was masked by the elevated pre-stress plasma cortisol levels in these groups.

Lactate, also used as a stress index, has been shown to increase in response to stress stimulus (Mesa and Schreck 1989, Gadomski et al. 1994). As with glucose and cortisol, blood plasma concentrations of lactate can rise dramatically with exposure to a stressor, returning to pre-exposure levels after several hours following suspension of, or acclimation to, the stimulus.

Blood plasma indicators have been used routinely as stress indices during the fish bypass system evaluation process for upgraded facilities constructed at FCRPS projects since 1990. At each dam, blood samples were collected from chinook salmon and steelhead smolts at successive points within the bypass system. Since blood plasma indicator levels in fish captured from the gatewell are similar to levels found in hatchery reared (naive) smolts (Congleton et al. 1984), results were compared to samples taken from smolts collected concurrently from gatewells. This process was replicated several times through the outmigration to accommodate timing differences. In all cases, blood plasma concentrations of cortisol, glucose, and lactate were measured as indices of stress resulting from passage through the facility. Mean levels of indicators (averaged across the entire outmigration) are included in Table 8 by facility for each species evaluated.

Results from these studies vary by site. For yearling chinook salmon, there were significant increases in cortisol and glucose concentrations at Little Goose (Monk et al. 1992b), Lower Monumental (Marsh et al. 1995) and John Day (Absolon et al. in prep.) Dams as fish passed from the gatewells through the sample tanks, while no significant differences were reported for any indicator levels from Ice Harbor (Gessel et al. 1997) and McNary (Marsh et al. 1996) Dams. Lactate levels increased significantly only at Little Goose Dam. Subyearling chinook salmon, evaluated only at McNary Dam, showed a real change only for mean lactate values, which declined significantly before returning to gatewell levels in raceways.

Steelhead cortisol concentrations increased significantly through all bypass systems except at McNary Dam, and significantly elevated levels of plasma glucose were recorded at Little Goose, Lower Monumental, and McNary Dams. Mean lactate concentrations in steelhead plasma showed significant increases only at Little Goose and Lower Monumental Dams.

Table 8. Most recent juvenile chinook salmon (*Oncorhynchus tshawytscha*) and steelhead (*O. mykiss*) mean blood plasma stress indicator concentrations by juvenile fish facility sample location for COE operated hydroelectric projects on the Columbia and Lower Snake Rivers.

					Blood plasma indicator concentration		
Hydroelectric Project	Evaluation year	Authority	Species	Sample location	Cortisol (ng/ml)	Glucose (mg/dl)	Lactate (mg/dl)
John Day	1998	Absolon et al. (in prep.)	yearling chinook salmon	gatewell	103.2	63.2	91.2
				pre-separator	151.6	76.2	82.6
				pre-sample tank	160.1	73.2	85.7
			steelhead	gatewell	98.8	88.2	98.6
				pre-separator	192.7	81.4	89.3
				pre-sample tank	179.0	107.5	105.5
McNary	1994	Marsh et al. 1996	yearling chinook salmon	gatewell	92.5	93.9	62.1
				post-dewaterer	97.5	97.1	74.8
				separator	102.4	91.6	65.3
				raceway (0-hour)	89.5	87.7	66.9
				raceway (2-hour)	98.4	93.1	61.5
				raceway (4-hour)	84.7	86.2	57.1
				raceway (6-hour)	79.8	88.3	52.3
				raceway (10-hour)	92.3	103.2	61.7
			steelhead	gatewell	83.2	153.3	59.3
				post-dewaterer	84.4	108.2	49.9
				separator	90.5	109.7	59.9
				raceway (0-hour)	101.2	132.7	72.4
				raceway (2-hour)	108.0	137.3	62.4
				raceway (4-hour)	86.9	132.7	59.6
				raceway (6-hour)	75.8	131.2	53.4
				raceway (10-hour)	100.9	161.5	62.6

Table 8. Continued.

Hydroelectric Project	Evaluation year	Authority	Species	Sample location	Blood plasma indicator concentration Cortisol (ng/ml)	Glucose (mg/dl)	Lactate (mg/dl)
McNary	1994	Marsh et.al 1996	subyearling chinook salmon	gatewell	57.2	92.7	122.8
				post-dewaterer	82.6	85.1	75.5
				separator	75.9	88.2	77.7
				raceway (0-hour)	84.4	73.1	114.2
				raceway (2-hour)	70.8	82.0	82.0
McNary	1994	Marsh et al. 1996	subyearling chinook salmon	raceway (4-hour)	75.6	80.9	78.4
				raceway (6-hour)	68.1	87.0	77.3
				raceway (10-hour)	88.3	89.2	87.0
Ice Harbor	1995	Gessel et al. 1997	yearling chinook salmon	gatewell	140.3	105.9	53.7
				pre-separator	140.7	95.2	59.2
				pre-sample tank	135.9	97.2	60.2
			steelhead	gatewell	101.0	131.4	51.4
				pre-separator	165.5	117.7	71.1
				pre-sample tank	188.8	112.9	65.2
Lower Monumental	1993	Marsh et al. 1995	yearling chinook salmon	gatewell	115.7	62.5	114.5
				post-dewaterer	144.6	68.9	73.4
				pre-separator	144.7	81.5	66.8
				raceway (0-hour)	152.3	103.1	57.9
				raceway (2-hour)	127.2	110.4	49.0
				raceway (4-hour)	121.4	111.7	50.4
				raceway (6-hour)	110.3	108.3	55.9
				raceway (10-hour)	166.6	104.3	57.5
			steelhead	gatewell	84.4	119.8	54.6
				post-dewaterer	154.2	116.3	60.6
				pre-separator	184.0	127.0	72.7
				raceway (0-hour)	138.6	147.0	52.1

Table 8. Continued.

Hydroelectric Project	Evaluation year	Authority	Species	Sample location	Blood plasma indicator concentration Cortisol (ng/ml)	Glucose (mg/dl)	Lactate (mg/dl)
				raceway (2-hour)	173.1	147.2	44.9
				raceway (4-hour)	69.3	149.7	55.7
				raceway (6-hour)	103.2	138.0	54.8
				raceway (10-hour)	174.1	142.8	59.8
Little Goose	1990	Monk et al. 1992b	yearling chinook salmon	gatewell	75.7	96.5	56.4
				post-dewaterer	77.3	87.5	75.1
				pre-separator	112.0	86.7	72.6
				raceway (0-hour)	140.7	107.8	71.0
Little Goose	1990	Monk et al. 1992b	yearling chinook salmon	raceway (2-hour)	160.5	164.9	59.5
				raceway (4-hour)	129.4	155.6	55.7
				raceway (6-hour)	85.5	154.2	58.2
				raceway (10-hour)	81.7	160.4	72.2
				pre-barge	79.4	109.2	55.6
			steelhead	gatewell	42.2	133.0	36.6
				post-dewaterer	114.5	126.3	67.5
				pre-separator	142.3	119.5	74.8
				raceway (0-hour)	125.1	109.9	68.7
				raceway (2-hour)	91.2	150.1	44.7
				raceway (4-hour)	61.0	142.1	46.8
				raceway (6-hour)	65.3	129.1	46.3
				raceway (10-hour)	103.5	147.0	53.2
				pre-barge	125.4	140.8	51.4

In general, blood plasma index responses through current bypass systems are within the normal range of expected values, given the stimuli. In cases where levels were measured following the bypass process (e.g., from raceways), plasma indicator concentrations returned to near gatewell levels within a few hours in non-stressed fish.

Results to date suggest that physiological effects of passage through fish bypass facilities on juvenile chinook salmon and steelhead are nominal, as measured by blood plasma stress indices, and follow a typical sequence for fish subjected to a stressor followed by acclimation to or removal of the stress (Wedemeyer 1976, Mazeaud et al. 1977). Several researchers have reported that effects of stress as measured by hepatic plasma indicators diminish within 24 hours following removal of the stressor in a protected environment (Sharpe et al. 1988, Gadomski et al. 1994, Schreck and Knoebl 1997). For example, transported fish showed an initial increase in plasma cortisol levels immediately after loading onto a barge, but the effect abated after three hours to pre-loading levels (Maule et al. 1988).

The relationships between physiological indicators of bypass-induced stress and in-river survival are not as well documented. There is evidence that short-term survival may be directly impaired as a result of stress in poor quality chinook salmon smolts. Under controlled conditions, Barton et al. (1986) demonstrated that multiple acute disturbances, similar to those encountered during the bypass process, resulted in cumulative physiological responses. Plasma indicator levels of healthy animals returned to control levels within 30 hours with no mortality after 3 stress events, while analogous treatment for unhealthy fish resulted in 50% mortality within 3 to 6 hours after the third event.

Indirectly, bypass stress may also contribute to reduced ability to respond successfully to in-river conditions. Barton and Schreck (1987b) found a positive relationship between metabolic rate and the plasma cortisol level in stressed fish. They concluded that even relatively minor events can reduce available energy stores in fish by as much as one quarter, leaving the animal with substantially fewer reserves to cope with environmental challenges such as temperature adaptation, disease, and demands on swimming stamina.

There is evidence that weaker fish may be targeted differentially for predation as a result of predator selection or prey vulnerability. For example, Congleton et al. (1984) demonstrated a significant positive correlation between impaired predator avoidance and crowding stress resulting in plasma cortisol levels of 75-150 ng/ml. In another study, Gadomski et al. (1994) could not find a significant relationship between juvenile salmonid descaling and predation by northern squawfish, but noted there was evidence of increased predation of descaled fish compared to controls at higher descaling rates. The authors suggested that under field conditions, the stress related to descaling may be more serious than the injury itself, resulting in decreased performance and greater susceptibility to predation.

JUVENILE PASSAGE THROUGH SURFACE BYPASS SYSTEMS AND SLUICEWAYS

Introduction

Prototype and permanent surface bypass and collection systems are under assessment at numerous federal and public mainstem Columbia and Snake River hydroelectric dams. This passage concept is based on successful development of a permanent surface collection/bypass system at Wells Dam between 1981 and 1989 (Johnson et al. 1992), and three additional years of evaluation to substantiate spring and summer juvenile migrant passage performance.

Bypass efficiency (ratio of fish passing surface bypass to total passage at test units) was 89% during that period (Dauble et al. 1999). The concept was developed based on the working premise, discussed below, that if smolts had the opportunity to discover a near-surface passage route, they would prefer this to sounding to pass through turbine intakes. The previous indications that surface routes would pass surface-oriented juveniles were the evaluations of passage through ice and trash sluiceways at The Dalles, Ice Harbor, and Bonneville Dams (Christensen and Wielick 1995).

Based on the successful adaptation of the ice and trash sluiceway surface bypass principles into a surface bypass system at Wells Dam, prototype surface bypass/collectors have been tested at a number of locations in the Pacific Northwest since 1994, especially at lower Snake and Columbia River dams. The goal of the prototype testing at these sites has been to determine whether use of bioengineering principles observed at the Wells Dam surface bypass system can be successfully applied at dams with appreciably different and more typical powerhouse/spillway project configurations.

Wells Dam Surface Bypass System

Wells Dam is configured differently than all other run-of-the-river dams on the Snake and Columbia Rivers. It has deep powerhouse turbine intakes with 11 spillway gates located over and between the ten turbines. This unique structure is referred to as a hydrocombine. Turbine intake ceilings are 21.3 m (70 ft) deep. Each of five turbine pairs now has one surface bypass entrance 4.88 m (16 ft) wide and 22.25 m (73 ft) deep. The surface bypass entrance velocity is approximately 0.61 m/s (2 ft/s), and each surface bypass slot passes up to 62.3 m^3/s (2,200 cfs). The surface entrances are part of the surface-oriented bulkheads installed directly above turbine intakes and in front of each pair of spillbays. Bypassed fish and flow pass through the surface entrances into an afterbay between the slotted entrance and the spill gates, and are discharged to tailwater through the spillway lift gates. Spill gates are the controls that dictate surface entrance flow and velocity. Surface bypass flow is 5 to 7% of hydraulic capacity for each turbine pair (Johnson 1996).

Bypass entrance slot guidance efficiency (ratio of fish passing into bypass slot relative to fish passing both slot and turbine pair) has been approximately 90% (Johnson et al. 1992), ranging from 84.3 to 95.0% for spring migrants and 76.5 to 97.0% for summer migrants (Skalski et al. 1996).

Surface Bypass Premises

Based largely on what was learned about juvenile behavior in the Wells Dam forebay, biological premises that established testable hypotheses were developed (Dauble et al. 1999) that provide a conceptual framework for the application of surface bypass concepts to prototype facilities at other locations:

1) Smolts follow the bulk flow as they approach a dam.

Forebay hydraulic conditions are influenced by river discharge, bathymetry, and turbine/spill operations These variables affect fish approach patterns and influence relative numbers of smolts available for spill and other passage routes. Juvenile radiotelemetry and hydroacoustic studies show that smolt densities are usually higher in the main channel than shoreline areas. Thus, most smolts follow the bulk flow in the thalweg (Johnson et al. 1999).

2) Smolts can discover a surface bypass flow net.

The horizontal and vertical distribution of smolts must be considered when locating surface bypass entrances to maximize the smolts' "opportunity for discovery" of the attraction flow net provided by the surface bypass entrance. For a partial powerhouse prototype, both the vertical and horizontal components of smolt distribution must be considered in the design of prototype surface bypass systems. For a full powerhouse design, like Wells Dam where the surface bypass system covers the entire powerhouse, only the vertical component needs to be satisfied to be successful. The design features of entrance size and number must work together to increase the opportunity for smolts to discover and use entrance flow fields projecting upstream into the forebay (Rainey 1997).

3) Surface bypass entrance conditions should not elicit an avoidance response.

Milling behavior, lateral movement, and upstream movement of smolts has been observed in evaluations of prototype surface bypass systems (Johnson et al. 1998) . Hydraulic conditions that enhance use of entrances are thought to include no abruptly decelerating velocity or "null zones," weak downward currents, or localized, intense water-particle acceleration fields. Brett and Alderdice (1958) found that improved bypass efficiency resulted from uniform acceleration of water velocities and minimization of visual and turbulence cues within the bypass entrance.

The relative importance of light and sound to surface bypass entrance conditions and how these variables influence fish behavior is unknown. However, the ability of fish to respond to local turbulence and velocity components of flow fields is well documented. For example, recent studies at Columbia River dams indicate that smolts detect and respond to near-field flow characteristics associated with different bypass screen designs (Nestler and Davidson 1995). Haro et al. (1998) suggested that bypass entrances need to be large enough to accommodate large schools of fish in order to be effective. This indicates that flow nets may have a threshold size, below which smolt passage is reduced.

4) Smolts stay in and pass through the conveyance structure safely.

Limited data support the premise that a powerhouse surface bypass with a high volume channel connected to a spillway can quickly and safely deposit smolts at a spillbay outfall (Adams et al. 1998).

5) Smolts safely enter the tailrace and quickly migrate downstream.

The benefits of efficient collection and safe conveyance is negated if survival rates upon egress are low. Indirect data from radio-tagged juveniles, and direct data from radio-tagged predators (northern pikeminnow) indicate that smolts safely entered the tailrace and quickly migrated downstream at Lower Granite Dam (Piaskowski et al. 1998).

Surface Bypass Designations

During the mid- and late-1990s, evaluations of prototype surface bypass systems have occurred or will occur in the future at Bonneville (both powerhouses), The Dalles, John Day, Ice Harbor, Lower Granite, Wanapum, Rocky Reach, and Wells Dams. Based on these prototypes, five basic surface bypass designations characterize the surface bypass systems tested to date:

1) Powerhouse Surface Flow Attraction Channel
2) Powerhouse End Collector
3) Surface Bypass Spill/Sluice
4) Occlusion
5) Hybrid

Some prototypes fall into more than one designation, and are cross-referenced.

Powerhouse Surface Flow Attraction Channel

This designation applies to a surface bypass channel with one or more entrances on the upstream face of a powerhouse. This emulates the successful permanent facilities at Wells Dam. Three prototypes fit this category: Lower Granite Dam, Bonneville First Powerhouse, and Wanapum Dam. The plan for each was to first determine whether adequate numbers of fish could be collected from the forebay, and then determine how to safely route flow either to the tailrace, or to dewatering and transportation facilities.

Lower Granite--Lower Granite Dam has a six-turbine powerhouse located immediately to the south of eight spill bays. The prototype configuration included a Surface/Bypass Collector (SBC) immediately upstream from and partially occluding, turbine Units 4 to 6. Attraction flow passes into one or more of four adjustable entrances and is conveyed to the adjacent spillbay through a longitudinal channel. The spillbay's Tainter gate provides flow-control for the prototype. Deep, wide entrance configurations (similar to Wells) were tested in 1996-1998. Tests in 1998-2000 were conducted with a Behavioral Guidance Structure (BGS) and simulated Wells Dam intake to control horizontal and vertical distribution, respectively, and increase collection. Also, in 1999 the emphasis was on enhancing guidance through greater surface-oriented entrance velocity (Anglea 1999).

In 1998, based on hydroacoustic estimates (Johnson et al. 1998), 36% of all outmigrants that approached the dam passed through the collector. Of those that passed through or below the prototype (through turbine intakes at Units 4 to 6) 51% entered the collector. In 1998, based on radiotelemetry estimates (Adams et al. 1998), 28, 29, and 49% of the wild steelhead, hatchery chinook, and hatchery steelhead, respectively, entered the collector of those fish that either entered the collector or passed through Units 4 to 6. In 1999, based on hydroacoustic estimates, 57.8% entered the collector that passed through or below the collector (through turbine intakes at Units 4 to 6) (Anglea 1999). Radiotelemetry evaluations were not conducted in 1999. The disparity between the hydroacoustic and radiotelemetry estimates may be due to the abundance of hatchery steelhead observed in the hydroacoustic samples, which are not species specific.

Adams et al. (1998) found that 54% of the radio-tagged subyearling fall chinook salmon approaching the powerhouse utilized the collector in 1998. In 2000, radiotelemetry and hydroacoustic evaluations of the surface collector continued. Results of both methodologies suggested that while the surface collector appeared to effectively pass fish relative to similar systems at other dams, it may not function as a stand alone surface bypass structure (Anglea et al. 2001; Plumb et al. 2002). In addition, the BGS appeared to effectively divert fish away from turbine units 1-3 and toward the SBC, but at too great a depth for high numbers of fish to effectively enter the bypass structure.

In 2001, in an effort to further improve fish passage at the dam, a Removable Spillway Weir (RSW) was installed next to the powerhouse in spillway 1 and was used in conjunction the SBC and BGS to improve bypass efficiency. A preliminary study conducted using balloon-tagged hatchery spring chinook salmon released directly into the RSW in November showed that survival and injury rates were similar for fish passing through the RSW and those passing through a spillway with flip lips (Normandeau Associates et al. 2002).

In 2002, radio tags and hydroacoustics were used to evaluate the RSW as a surface-flow bypass tool. Information from both methods indicated that the RSW was an effective surface-flow fish bypass system for both spring chinook salmon and steelhead (Anglea et al. 2003; Plumb et al. 2003). Also in 2002, physical and biological data were integrated by simultaneously conducting three-dimensional tracking of fish (using acoustic tags) and numerical modeling of the hydraulic flow fields (Cash et al. 2003). This will allow a greater understanding of the behavioral response to flow velocity and acceleration and have broad application to many surface bypass systems at various dams.

Further testing using radio tags in 2003 indicated that the RSW likely reduced passage times, and also passed higher percentages of fish than all other routes while using less water (Plumb et al. 2003). The authors found no differences in survival probabilities between conditions tested, suggesting the RSW posed few adverse effects on survival relative to gas cap spill.

Bonneville First Powerhouse --The Bonneville First Powerhouse consists of ten turbines and is separated from the spillway by Bradford Island. A prototype surface collector in front of turbine Units 3 to 6 was installed in 1998 and evaluated to determine whether occluding the upper intake and passing high flows through a deep slot of variable width would successfully attract fish. The bottom of the collector was 36 to 42 ft deep. Both 5- and 20-ft slot widths were tested, passing approximately 1,000 cfs and 3,000 cfs, respectively. Flow and fish passing each slot were routed into the upper turbine intake.

Ploskey et al. (1999) evaluated the prototype collector and found effectiveness was 89 to 90% for both spring and summer migrants and both slot widths, based on committed fish that had passed into or under the prototype collector.

Hensleigh et al. (1998) found that approximately 45 and 40% of the radio-tagged juvenile steelhead and yearling chinook salmon, respectively, that were contacted upstream from the Bonneville First Powerhouse were contacted at the prototype collector. A total of 67% of the radio-tagged fish that contacted the prototype collector did not enter, and instead moved south along the face of the structure and initially held upstream from turbine Units 1 to 2.

Tests in 2000 will include expanding the collector to include turbine Units 1 to 2 to address the question of fish that moved laterally in 1998. The permanent collector would tentatively have one entrance slot per turbine unit and bypass a total of 30,000 cfs. A decision to proceed with further development has tentatively been scheduled for 2000 or 2001, once the feasibility of passing juveniles at higher outfall volumes directly to the Bonneville First Powerhouse tailrace has been determined. Uncertainties associated with mortalities, both direct (from turbulent shear and strike) and indirect (from juvenile disorientation and predation), remain to be addressed. The option of a hybrid surface bypass collector, which utilizes an enlarged ice and trash sluiceway with an acceptable bypass outfall location along with ESBSs, has been discussed.

Wanapum--In 1995, Grant County Public Utility District installed and tested a 55-ft deep prototype at Wanapum Dam immediately upstream from turbine Units 7 to 10. A single 16-ft wide by 50-ft deep slot entrance was installed above Unit 8, with a hydraulic capacity of 1,400 cfs (Sverdrup Corporation 1995). In 1996 the prototype was extended to occlude turbine Units 4 to 6, and extended again in 1997 to occlude turbine Units 1 to 3.

Mean collector effectiveness for the single unit with an entrance (Unit 8) was 30% in 1995 (Ransom et al. 1995). However, efficiency decreased when Units 7, 9, and 10 were included. In 1995, the horizontal distribution of passage through turbines was skewed toward turbine units not covered by the occlusion device, as compared to previous years before the occlusion was installed (Ransom et al. 1995). Further, spillway effectiveness was higher from 1995 through 1997 (43%) than in previous years (30%) (Kumagai et al. 1996). Effectiveness goals were not met with the modification and extension of the collector in 1996-1997, and further testing was dropped after the 1997 passage season. No additional surface bypass evaluations are scheduled for Wanapum Dam.

Powerhouse End Collector

Powerhouse end surface bypass collector sites are unique since large numbers of juvenile outmigrants either pass or accumulate near the respective powerhouse collector locations. Each is characterized by a strong component of surface-oriented lateral flow that helps concentrate juvenile outmigrants. Examples include the Bonneville Second Powerhouse sluice chute, The Dalles Dam ice and trash sluiceway, and Rocky Reach Dam.

Bonneville Second Powerhouse Sluice Chute --The sluice chute is located immediately south of turbine Unit 11, the southern-most turbine at the powerhouse. The unique, excavated forebay bathymetry results in hydraulic conditions at the upstream powerhouse face that include strong lateral flows toward the north and south end of the powerhouse (Monk et al. 1999a), which concentrate juvenile fish at these locations.

In 1998, with the six southern-most turbine intake extensions removed, radiotelemetry studies detected 52% of steelhead and 36% of yearling chinook salmon moving through the sluice chute when open. Additionally, 21% of the steelhead and 14% of the yearling chinook salmon were detected in the juvenile bypass system. With the chute closed, 50% of the steelhead and 30% of the tagged yearling chinook were detected in the bypass system (Hensleigh et al. 1998). Ploskey et al. (1999) using hydroacoustics found the combined effectiveness of the sluice chute and FGE in turbine Units 11-13 was 90% during spring and summer. However, when the chute was closed, FGE of turbine Units 11-13 alone was 55% in the spring and 30% in the summer.

The effectiveness of the sluice chute, relative to total passage at turbine Units 11-13 plus passage at the sluice chute, averaged 83% in spring and 81% in summer. Efficiency was high; about five times more fish passed through the sluice chute than would be expected from the proportion of water passing the chute. The researchers concluded the sluice chute has great potential as a corner collector.

The guidance performance of the sluice chute appears quite promising. However, the current location of the outfall, which discharges chute flow into the immediate tailrace of the Bonneville Second Powerhouse, has been questioned due to poor tailrace conditions (Ledgerwood et al. 1990, 1991, 1994).

The Dalles Dam Ice and Trash Sluiceway--The Dalles Dam powerhouse is oriented parallel to the forebay river channel centerline. There is a surface-oriented, westerly lateral flow along the face of the powerhouse. The ice and trash sluiceway is located along the entire length of the 22-turbine-unit powerhouse. The ice and trash sluiceway is operated to pass up to 5,000 cfs (the hydraulic capacity of the sluiceway) through three or more chain gates. The priority chain gate operation is above turbine Unit 1 at the west end of powerhouse. During periods of no spill, sluiceway passage was estimated at 40 to 55% (Giorgi and Stevenson 1995).

In 1998, hydroacoustic estimates of spring passage were 40.7 and 25.8%, for 30 and 64% spill, respectively; summer passage was estimated to be 35.2 and 26.2%, for 30 and 64% spill, respectively (BioSonics Inc. 1999c) Preliminary results from hydroacoustic studies in 1999 indicate sluiceway passage during the spring was 15 and 12%, and during the summer was 9 and 10%, for 30 and 64% spill, respectively (Ploskey et al. 1999). Ploskey et al. (1999) used different techniques to estimate sluiceway passage than was used in previous years. In 1997, Hensleigh et al. (1999) found that the greatest number of observations of radio-tagged wild and hatchery steelhead and yearling and subyearling chinook salmon was at the west end of the powerhouse.

Substantial predation by northern pikeminnow (*Ptychocheilus oregonensis*) and gulls (*Larus* spp.) may occur at the sluiceway outfall and in the reef and island areas just downstream from the outfall, based upon northern pikeminnow abundance and stomach content evaluations (Hansel et al. 1993, Ward et al. 1995, Jones et al. 1997, and Snelling and Mattson 1998). Dawley et al. (1999a) found that survival (relative to tailrace reference releases) for sluiceway passage at The Dalles Dam in 1998 was 96 and 89% for coho and subyearling chinook salmon, respectively. These survival estimates did not differ appreciably from that of spillway passage at 30% spill.

Rocky Reach Dam --The Rocky Reach powerhouse is oriented parallel to the river channel centerline while the spillway is upstream and perpendicular to the forebay channel. Fish accumulate at the downstream half of the powerhouse (Dauble et al. 1999).

A collector with entrances above turbine Units 1 and 3 was tested in 1998. The Unit-1 entrance was tested with 7- and 15-ft wide by 56-ft deep slot entrances, which passed 2,275 and 3,000 cfs, respectively. Bypassed flow is routed to a dewatering screen and tailrace outfall. The Unit-3 entrance routes approximately 2,400 cfs into turbine Unit 2 where intake screens guide fish from the Unit 3 surface bypass slot upward into powerhouse gatewells. Entrance flow from both surface bypass slots passes through an intake venturi that for turbine Unit 1 is an occlusion device. Intake guidance screens in turbine Units 1 and 2 guide some fish that pass under the surface bypass facilities into powerhouse gatewells. All fish from the surface bypass entrances and powerhouse gatewells are conveyed to a common monitoring facility and outfall (Mosey et al. 1999).

Mosey et al. (1999) found that 27% of the PIT-tagged steelhead and yearling chinook salmon passed through the unit 1 and 3 surface bypass entrances, and 12 and 33% of the sockeye and subyearling chinook salmon, respectively, used the entrances in 1998.

Chelan County Public Utility District plans to continue development of and ultimately build a permanent collector system at Rocky Reach Dam, which will use a combination of slotted surface-oriented entrances and turbine intake screens to collect and bypass fish.

Surface Bypass Spill/Sluice

Surface bypass spill/sluice facilities route juvenile fish directly to tailwater, rather than into bypass systems with dewatering and sampling or transportation loading facilities. Examples include the proposed skeleton bay concept at John Day Dam, a Wanapum Dam sluice chute, a raised spillbay crest at Lower Granite and John Day Dams, and the RSW at lower Granite Dam. The design is based on the hypothesis that a strong, surface-oriented flow field projecting upstream into the forebay will provide an opportunity for juvenile salmon to discover and use these surface bypass routes.

John Day skeleton bay surface bypass--John Day Dam has 20 spillbays, 16 operating turbines, and 4 turbine skeleton bays between the powerhouse and spillway. The COE has developed a preliminary design of a single skeleton bay surface-oriented sluiceway that will pass approximately 18,000 cfs through three, 20-ft-wide ogee chutes with crests at elevation 243 feet msl (normal forebay elevation is 264 feet msl). Each chute would transition to a 20-ft horizontal apron at the downstream end of the ogee curve-to-tangent location. Control gates would operate either fully open or closed. The design memorandum has been completed. Radiotelemetry studies show that as many as 75% of juveniles pass the skeleton bay within 100 m of the proposed location (COE 1998d).

John Day spillbay raised-crest surface bypass--A spillbay raised-crest option is considered a less costly but potentially equally effective alternative to the skeleton bay concept, discussed above. Scoping investigations suggest raising the crest of Spillbay 20 will enable a surface spill of 14,000 cfs. The spillbay Tainter gate would be operated fully open or closed. The current plan is to prototype test this option in 2002.

Lower Granite spillbay raised-crest surface bypass--This option is similar to the John Day Dam spillbay raised-crest surface bypass. A strong lateral flow from the north occurs during zero or low spill periods. Also, it is hypothesized that fish that accumulate upstream from turbine Units 4 to 6 will detect the surface flow field and pass through the raised-crest entrance. Raised-crest flows are estimated at 6,000 cfs. A removable raised-crest prototype is scheduled for testing in 2001 at Spillbay 1. The prototype surface collector in front of the powerhouse will be partially removed, but the simulated Wells intake and behavioral guidance structure would remain to enhance the performance of the raised crest prototype.

Wanapum surface sluice chute--A surface sluice chute is located adjacent to Spillbay 12, consists of a 20-ft-wide by 10-ft-deep gated notch in the non-overflow wall, and has a hydraulic capacity of 2,000 cfs. Mean chute passage was 6.5% of the total outmigration from 1986 to 1996, in 1.6% of the total flow (Ransom 1997).

Occlusion

An important goal of surface bypass systems is to reduce fish entrainment into turbine intakes. Occlusion devices increase the turbine flow-field intensity near the channel bottom while reducing the flow-field intensity at mid-depth, allowing mid-depth fish to discover a surface bypass system entrance. The surface flow attraction channels and behavioral guidance structures described above are considered occlusion devices, as are blocked trashracks.

The Dalles Dam blocked trashracks--The Dalles Dam turbine intake ceilings are at elevation 145 ft msl, relative to a normal operating pool elevation of 160 feet msl. Blocking the upper 45 ft of intake trash racks at the west end of the powerhouse down to elevation 100 ft msl may reduce turbine entrainment and increase ice and trash sluiceway passage. In 1996, hydroacoustic studies showed no effect from the blocked trashrack

(BioSonics Inc. 1997). However, tests of a similar structure at Bonneville First Powerhouse in 1996 showed positive results (Ploskey et al. 1998). Tests at Lower Granite Dam in 1998 showed that higher surface bypass guidance could be achieved through use of the simulated Wells intake, which occludes the upper turbine intake with a blockage that extends 20 ft upstream and causes the turbine to draw flow from the lower forebay depths. The occlusion concept will be tested at The Dalles Dam in 2001.

Lower Granite Dam behavioral guidance structure and simulated Wells Dam intake --These structures worked in tandem in 1998 to reduce the number of fish approaching turbine Units 1 to 3 (horizontal occlusion) and reduce entrainment into turbine Units 4 to 6 (vertical occlusion). Since only the three southernmost turbines (Units 1 to 3) were occluded, flow deflection angles relative to the behavioral guidance structure were small. This was considered an important criterion by the design team, which was concerned that greater deflection angles would result in greater curtain tilt from vertical inclination and reduced effectiveness.

Dauble et al. (1999) found that 80% of the fish approaching the behavioral guidance structure and turbine Units 1 to 3 passed elsewhere. They concluded that fish did not guide along the guidance structure but tended to stay away from it. Fish milling behavior was reduced from the entire powerhouse near-forebay to the near-forebay zone upstream from the prototype surface collector. This suggests that a guidance structure could also improve spillway effectiveness and efficiency.

The simulated Wells intake was not assessed independently of the guidance structure since it could not be moved to allow a blocked, randomized test sequence. Adams and Rondorf (1999) found nearly a ten-fold increase in detections of radio-tagged fish near the dam in 1998 relative to tests conducted in 1996-1997. In 1998 fish appeared to meander more before passing, suggesting fish have more opportunities to discover surface bypass entrances prior to sounding into turbine intakes. The COE plans to test a 6,000 cfs surface discharge over a raised spillbay crest with a behavioral guidance structure and/or simulated Wells intake in 2001.

Wanapum Dam --This configuration is discussed above.

Rocky Reach Dam occlusion–This occlusion device was not evaluated separately from the surface flow prototype collector. However, it reduced turbine intake velocities near the intake ceiling and increased deeper turbine intake velocities. In 1996, an upstream-projecting floor-like shelf at the surface bypass slot entrance invert elevation was also installed at turbine Units 1 to 2 in an effort to reduce fish sounding and turbine entrainment. However, it was concluded that many fish still sounded upstream from the shelf (Steig and Adeniyi 1997).

Hybrid

Surface bypass alternatives continue to be developed in combination with existing project facilities such as spillways, sluiceways, and mechanical screen bypass systems to improve non-turbine passage. Examples of hybrid systems under consideration or development include the Bonneville First Powerhouse surface collector and ESBSs, The Dalles Dam turbine intake occlusion blocks with the ice and trash sluiceway, Lower Granite Dam surface collector, guidance structure, and simulated Wells intake with ESBSs, Lower Granite Dam spillbay raised-crest surface bypass and guidance structure, and the Rocky Reach Dam surface bypass slots with occlusion and turbine intake screens.

Surface Bypass Discussion

To date, surface bypass system prototypes have performed with mixed results, ranging from poor (Wanapum Dam) to good (Bonneville Dam and the RSW at Lower Granite Dam). Most of the prototype testing conducted to date has focused on entrance conditions and efficiency, Premises 1, 2, and 3, discussed above. Judging the overall effectiveness of surface bypass systems will have to include consideration of survival through the conveyance (including dewatering) and outfall/tailrace environments, Premises 4 and 5. Only then can their true efficacy be evaluated and compared to other passage routes.

JUVENILE PASSAGE THROUGH TURBINES

Background

Early studies of fish passage at FCRPS dams reported high levels of mortality for fish passing through turbines (Holmes 1952, Schoeneman et al. 1961, Long et al. 1968). As a result, fisheries agencies have since focused primarily on providing safe, non-turbine passage routes for juvenile fish as a means to improve survival. Nevertheless, substantial numbers of juvenile fish continue to pass through turbines, and reducing turbine-related mortality will improve the direct survival of fish migrating through the hydropower system. This reduction in mortality can be accomplished through modifying turbine operations, and possibly by installing minimum gap runner (MGR) turbines or developing new turbine designs that further improve the environment in the turbine through which fish pass.

Since 1995, turbines at Corps projects have been operated within 1% of peak unit efficiency during the juvenile outmigration period (NMFS 1995). Since 1997, the Corps has been conducting biological studies of factors affecting fish passage through mainstem Snake and Columbia River turbines and engineering studies of the hydraulic environment within these turbine through their Turbine Survival Program to reduce mortality associated with turbine passage (USACE 2003).

Recent Estimates of Survival Through Turbines

Recently, many estimates of turbine passage survival have been made using balloon-, PIT-, and radio-tag methodologies. Each method has strengths and weaknesses, but compared to early studies of dam passage survival based on batch marks, all provide increased rigor and precision. In fact, today it is not uncommon to combine two of the methods to address a specific question or research objective. Recent estimates of juvenile fish survival through turbines are summarized in the Turbine Survival Program summary report (USACE 2003). Here we report unpublished preliminary results from a NOAA Fisheries evaluation of survival at Ice Harbor Dam in 2003, and provide some observations on estimates of turbine survival using the various mark-recapture methods.

At Ice Harbor Dam in 2003 NOAA Fisheries released PIT tagged run-of-river hatchery yearling chinook salmon into the bypass collection channel, spillway, turbine unit slots 1A and 3A, and a control site 1 km downstream. Turbines during the study were operated normally, within the 1% range. Point estimates of relative survival were 98, 88, and 88% for the bypass, turbine 1A, and turbine 3A releases, respectively. Point estimates of relative survival for run-of-river hatchery subyearling chinook salmon were 99, 97, and 87% for the bypass, spillway, and turbine 1A releases, respectively.

Absolon et al. (2002) estimated that relative survival of PIT-tagged yearling chinook and coho salmon through turbines (turbine releases compared to tailrace releases made approximately 2 km downstream from the project) was 79.0 and 83.0% at The Dalles Dam in 2000 during the daytime and evening, respectively. The relative survival of subyearling chinook salmon passing through turbines during the summer was estimated at 79.1 and 88.9% during the daytime and evening, respectively. This compares with an estimated yearling chinook salmon survival through The Dalles Dam turbines of 86.9% in 2000, based on radiotelemetry (Counihan et al. 2002).

While the differences between the PIT- and radio tag-based estimates during the spring are not likely significant, the trend was for the radio-tag estimates to be higher than the PIT estimates. Also, the PIT tag nighttime estimates during the summer were significantly higher than estimates made from daytime releases, suggesting a possible predation component exists for fish passing through turbines when the environment of the immediate tailrace is captured in the treatment.

At John Day Dam evaluations of survival through turbines were conducted in 2002 and 2003 using radiotelemetry (Counihan et al. 2002, 2003). Test were conducted during both day and night under different spill conditions each season and year, confounding any comparison between day and night estimates similar to those discussed above for The Dalles Dam. For yearling chinook salmon, relative survival ranged from 76.4 to 83.2% for both years, and for subyearling chinook salmon in 2003 mean relative survival was 72.9 and 72.2% for the two spill treatment tested.

In the Lower Snake River, Muir et al. (2001) estimated that relative survival of PIT-tagged yearling salmon passing through turbines at Lower Granite, Little Goose, and Lower Monumental Dams was 93, 92, and 87%, respectively. Finally, Bickford and Skalski (2000) conducted an analysis of 102 replicate releases of salmonid smolts migrating through Snake and Columbia River Dams and estimate that average survival through Kaplan turbines is 87.3%.

Operation of Existing Turbines

Turbine efficiency is considered to have a relatively direct effect on fish passage survival. Inefficient turbine operation is a result of a poor blade-to-wicket gate relationship, where efficiency drops due to turbulence and vortex shedding as a result of the rotating machinery (hub and blades) being misaligned with the hydraulic flow field coming off the stationary, but adjustable, wicket gates. The relationship between survival of juvenile fish passing through Kaplan turbines is thought to be positively correlated with unit efficiency, based largely on a visual interpretation of the figures provided by Oligher and Donaldson (1966). However, a statistical evaluation of the same data sets would likely produce a low correlation between unit efficiency and fish survival. Bell et al. (1981) reviewed all the available data and recommended making every effort to operate turbines at peak efficiency at a given head during periods of peak fish passage to minimize fish mortality.

All mainstem FCRPS turbines are vertical-axis Kaplans and all have six blades, except for Bonneville Dam turbines, which have five blades due to the lower head. The Kaplan design allows the blade angle to be adjusted (the blade rotated) up or down to best fit the incoming flow, which varies with degree of wicket gate opening. Wicket gates are adjusted to control flow through the machine, and thus power output for a given head. For any given head, Kaplan turbines can operate over a wide range of power outputs and flow, and thus maintain a high level of efficiency over a broad range of operating conditions (head and load). Curves relating efficiency to unit performance (power output or flow) are developed for each head, and since head varies with forebay and tailwater elevation, a three-dimensional "hill" diagram of unit efficiency can be developed.

NOAA Fisheries recognized the relationship between turbine efficiency and fish survival and to protect salmon stocks listed under the Endangered Species Act, adopted this recommendation as part of their Biological Opinions (NMFS 1995, 2000). Today, turbines at all eight mainstem Corps projects are operated within 1% of peak efficiency. The Corps provides the turbine operating points necessary to meet the BiOp requirement in their annual Fish Passage Plan, and any updates are coordinated with Bonneville Power Administration and salmon managers through the Corps Fish Facility Operation and Maintenance Committee.

Juvenile salmon mortality associated with passing mainstem Columbia River turbines is potentially caused by several mechanisms (Ferguson 1993, USACE 1995, Cada et al. 1997). The turbine environment through which juvenile salmon pass during their downstream migration varies physically with load, head, runner speed, and fish location within the intake. It also varies over time with environmental conditions such as temperature and fish condition. While efforts are underway to explore the relationships between these various factors and fish survival (for example, Carlson and Duncan. 2003), turbines are currently operated within 1% of peak efficiency based on the assumption that fish survival and hydraulic efficiency are correlated.

Operating near peak efficiency offers project operators simple guidelines that are convenient for management purposes, but are not linked to any rigorous assessments of the potential relationship between unit efficiency and fish survival. Indeed, results of studies undertaken recently to explore this relationship question the basic assumption that peak fish survival is associated with peak unit efficiency. Mathur et al. (2000) using yearling chinook salmon and balloon tags evaluated survival at Lower Granite Dam. They found no significant differences ($P > 0.05$) between survival estimates for fish passing through a turbine operated at the low end of the 1% efficiency range, near peak, and a high flow level that caused cavitation, and maximum survival occurred at the low end of the 1% range.

Normandeau Associates et al. (2000) evaluated smolt survival through Bonneville First Powerhouse turbines under four discharge levels and at three different release locations within the turbine (tip, mid-blade, and hub). Survival varied more with fish distribution (release location) than discharge, and maximum survival did not occur at peak efficiency. Normandeau Associates et al. (1996a) evaluated smolt survival through Rocky Reach Dam turbines under three discharge levels and at two different release locations within the turbine (3.0 and 9.1m below the intake ceiling). Peak survival was observed to occur within the 1% range, but did not occur at peak efficiency. In contrast, in a similar study conducted at Wanapum Dam, Normandeau Associates et al. (1996b) concluded that maximum survival occurred outside of peak efficiency.

In 2002, NOAA Fisheries conducted an evaluation of fish survival through McNary Dam turbines. We evaluated the condition of yearling chinook salmon passing through turbine intake gatewells prior to entering the juvenile bypass, and the survival of radio tagged yearling chinook salmon passing through turbines and immediate 2 km of tailrace under two different operating conditions. One was a discharge of 317 m^3s^{-1} (11,200 cfs) and at the upper end of the 1% range from peak efficiency, and one was a discharge of 464 m^3s^{-1} (16,400 cfs) which is outside the 1% range and at the maximum "on cam" blade position based on Wittinger and Polinsky (2002).

For both test operations, the condition of fish guided into turbine intake gatewells by fish guidance screens and direct and indirect mortality associated with turbine passage were assessed. We found no differences in fish condition between the two gatewell environments tested, and descaling and injury rates were low. Fish travel time from time-of-release in the gatewell to time-of-detection at the juvenile bypass system PIT-tag detectors ranged from 0.07 to 136.6 hours (median 0.48 hours) for gatewell slot 9B operated at 16,400 cfs, and from 0.02 to 276.0 hours (median 18.7 hours) for gatewell slot 8B operated within 1% of peak efficiency. Relative survivals (comparison of turbine to tailrace releases) of test fish were 87.3 and 85.5% for the 11,200 cfs and 16,400 cfs operations, respectively ($P = 0.452$). Also, we observed twice as many radio tag detections on antennae located on a wing wall north of the spillway from fish released under the 11,200 cfs turbine operation compared to the 16,400 cfs operation. This indicates that unit operation affects flow patterns, fish behavior, and routing in the tailrace, which may in turn lead to differential indirect mortality from predation in the tailrace.

Our relative survival estimates for fish passing through McNary Dam turbines under the two operating conditions based on radiotelemetry are similar to recent estimates of turbine survival at other Snake and Columbia River dams, discussed above. Normandeau Associates et al. (2003) also found no significant difference in fish survival between the two operating conditions. They estimate that immediate or “direct” turbine survival (90% CI) was 93.0% (90.0-97.0%) and 94.6% (91.5-98.1%) at the 317 m^3s^{-1} and 464 m^3s^{-1} operations, respectively.

Skalski et al. (2002) completed the most rigorous review to date of the relationship between salmon survival and turbine operating efficiency. Their retrospective analysis concluded that at four Snake and Columbia River dams, peak passage survival did not occur at peak turbine efficiency. They found as much as a 3.2% difference in survival between the maximum survival and survival at peak turbine efficiency. Furthermore, a meta-analysis using results from 11 balloon tag survival studies conducted at different sites found no association between survival and efficiency ($r^2 = 0.0311$, $P = 0.2640$). They recommended that turbines be operated to maximize turbine passage survival, and not be based solely on peak operating efficiency. Their findings reflect the complex and variable nature of the relationship between fish survival and turbine operations.

Minimum Gap Runners

A minimum gap runner (MGR) is a Kaplan turbine design which minimizes the clear openings between the rotating blade tip and the stationary discharge ring, and between the base of the blade and the hub. The concept is to reduce these openings to levels which are technically reasonable without causing mechanical interference between moving parts, while maintaining reasonable turbine performance. This reduces the opportunity for fish to be entrained in high velocity discharges through these narrow gaps, which is potentially harmful to fish (Eicher Associates Inc. 1987, USACE 1995).

To minimize the gaps near the hub, the shape of the runner hub must be modified to conform to the turbine blades throughout their full range of movement. The simplest means of accomplishing this is to use a spherical hub in combination with a specially shaped runner cone which allows water passing the blades to exit with a constant velocity and without excessive turbulence. Closing the gaps at the hub reduces leakage and increases turbine efficiency, but the shape change of the hub and cone decrease efficiency if blades are adjusted very far from the optimal design conditions (Rod Wittinger, Portland District, COE, Pers. commun, August 1999).

Minimizing the blade-tip gap throughout the full range of the blade poses a different design challenge. For a standard machine, the blade periphery is cut in a spherical shape corresponding to a discharge ring, which is the fixed structure or wall that the rotating machinery moves within. The discharge ring is spherical below and conical above the turbine blade axis of rotation, allowing the turbine runner to be removed from above for maintenance. In the MGR, the spherical portion of the discharge ring extends upward above the axis of rotation of the runner and encompasses a greater portion, if not all, of the runner tip. This is accomplished by installing a spherical-shaped discharge ring which conforms to the shape of the runner tip after the turbine runner has been installed.

MGRs are being installed in all ten units of the Bonneville First Powerhouse. The efficiency curves of an MGR design are steeper and have more of a “peak” shape than the broad “hill” seen with standard Kaplan designs. Steeper efficiency curves mean that operating within 1% of peak efficiency for fish passage will narrow the operating range, and powerhouse “capacity” will be reduced.

Fish survival through a MGR turbine at Bonneville Dam First Powerhouse was tested during the winter of 1999-2000 using balloon tags (Normandeau Associates et al. 2000). The MGR was compared to a standard unit (Unit 5). The blade-to-wicket gate relationships for the standard unit were adjusted or “tuned” prior to the test. Survival of juvenile salmonids through the MGR turbines was evaluated for three different release locations within the turbine (tip, mid-blade, and hub), each tested under four different discharge levels: 176, 198, 297, and 340 m^3s^{-1}. Survival varied more with fish distribution (release location) than discharge. For both types of turbines, survival was higher for fish released near the hub and mid-blade locations than for fish released at blade tips.

Estimated survival for the MGR ranged from 96 to 101% and 96 to 98% for the hub and mid-blade releases, respectively. Estimated survival for blade-tip releases ranged from 91 to less than 96%. Maximum survival occurred at the peak unit efficiency for the blade tip and mid-blade releases, but for the hub releases maximum survival did not occur at peak efficiency. In fact, the lowest survival was at peak efficiency. Skalski et al. (2002) completed additional analyses of the Bonneville Dam MGR data and found no significant relationship between salmonid survival and turbine efficiency at any of the three release locations, nor between survival and average head, blade angle, discharge, or power generated.

Corps Turbine Passage Survival Program: Improvements to Increase the Survival of Fish Passing Through Kaplan Turbines

The Corps Turbine Passage Survival Program (TSP) was developed to quantitatively evaluate juvenile fish passage through Kaplan turbines with an emphasis on identifying turbine structures and operations responsible for injuring fish (USACE 2003). The Program has four primary objectives:

- Evaluate and recommend operational criteria to improve the survival of fish passing through turbines.
- Identify biological design criteria for modifying existing turbines.
- Investigate modifications to existing designs that may increase fish survival associated with turbine passage.
- Recommend turbine rehabilitation or replacement that incorporates improvements for fish passage survival.

The Program has been ongoing since 1997 and a significant amount of work on hydraulic model studies, engineering evaluations, and biological investigations has been conducted to relate the physical conditions fish are exposed to turbine operations and biological outcomes.

For example, Carlson et al. (2002) compared three-dimensional tracks of juvenile steelhead and chinook salmon with neutrally buoyant drogues passing through turbines at McNary Dam under two different operations. They found a significant difference in elevation component of trajectories and time to pass the turbine intakes between the juvenile salmon and drogues, and made recommendations on the use of physical models to simulate fish passage through turbines. They also conducted a sensitivity analysis between total fish survival through turbines and passage through the tip and hub areas of the turbine runner, and found that total survival through a turbine was relatively insensitive to the proportion of fish passing the tip and hub areas of the turbine runner.

Carlson and Duncan (2003) released sensor fish through turbines at McNary Dam in 2002 in conjunction with the Absolon et al. (2003) and Normandeau Associates et al. (2003). They observed several differences in the physical conditions fish were exposed to with the two different operations, including draft tube turbulence, time-to-pass the turbine runner, and acceleration impulses believed to be associated with the stay vane-wicket gate cascade. Although outside of the TSP, Coutant and Whitney (2000) conducted an literature-based evaluation to develop biological information normal behavioral patterns of fish for use in engineering designs to improve the survival of fish passing through turbines. In general, they urged caution in the use of neutrally buoyant beads in physical hydraulic models to simulate fish trajectories, and state there are practical limits to the measurement of fish and flows in the vicinity of the turbine runner environment.

A recent Program report summarizes in detail the accomplishments to date (USACE 2003). The report concludes that:

- The distribution of fish through a turbine has been fully defined.
- Route-specific areas of the turbine can be tested for impacts on direct survival.
- Blade tips pose a greater risk to fish than the mid-blade or hub areas.
- As discharge increases, so does the hydraulic performance of the draft tube. Turbulence through turbines and passage times of neutrally buoyant beads are higher at lower discharges.

- The most severe hydraulic conditions occur at the trailing edges of wicket gates and runner blades, within the hub "rope", and near the leading edge of draft tube splitter walls.
- Fish guidance screens affect turbine performance through head loss and by altering flow distribution within the unit.
- While some improvements in turbine operations have been made, error exists within many unit control systems, reducing certainty as to the true turbine operating conditions.
- A statistically valid relationship between turbine operating efficiency and fish survival does not appear to exist.
- The duration of fish passage is shorter through turbines than spillway stilling basins.
- The physical conditions fish experience passing through turbines is less than through spillway stilling basins.
- Physical model studies are an essential component of biological test designs and interpretations.
- The rate of bead strike on structures in a model is several times the rate of observed physical injury on fish passing through the same prototype structure.
- Uncertainty exists regarding the distribution of run-of-river migrants through turbines, and current release strategies may result in test fish being more uniformly distributed, vertically, as they pass through wicket gates.
- While mechanical injury is the largest component of direct mortality, indirect mortality may be a more significant component of total mortality than is currently assumed.

Department of Energy Advanced Hydropower Turbine System Program

Since 1994, the U. S. Dept. of Energy (DOE) has funded studies under the Advanced Hydropower Turbine Systems Program (AHTS) that support the development of advanced, environmentally friendly turbines with the goal of improving hydropower's environmental performance. The DOE AHTS has funded several studies and development of tools to explore the causal mechanisms of fish injury and mortality associated with passage through turbines.

Ploskey and Carlson (2003) modeled the probability of blade strike for the Bonneville Dam First Powerhouse turbines and compared these predictions to empirical data collected in 1999 and 2000 by Normandeau and Associates (2000). Blade-strike models predicted that the probability for injury from strike increases with decreasing unit discharge and increases with increasing distance from the center of the runner where the fish passes the blade. However, there was not a strong relationship between injury and discharge in the empirical data. They speculate that injury from blade strike could decline with increased discharge, but be masked by increases in injury from other causes. They point out that researchers' inability to assign causal mechanisms to observed injuries hinders being able to relate injury or mortality to discharge. They conclude that efforts to empirically determine fish distributions at the turbine intake trashracks, wicket gates, and runner blades are needed before models can be used more effectively to identify turbine operations and designs that reduce injury and mortality rates.

Weiland and Carlson (2003) evaluated the feasibility of using ultrasonic tracking and acoustic cameras to observe fish and neutrally buoyant drogues during passage through turbines at McNary Dam. They found that while range of detection was high (30.5m) for both technologies, spatial resolution was low and affected by noise within the turbine environment and would have to be improved to increase the utility of using these technologies.

Abernathy et al. (2001) examined the responses of fish acclimated and not acclimated to gas supersaturation to rapid pressure changes to investigate the relative importance of pressure as a source of turbine passage mortality and injury. They found that the level of gas supersaturation that causes acute gas bubble trauma varies among species, as does the frequency, type, and severity of injuries associated with turbine passage. They concluded that if gas saturation is not a problem, exposure of juvenile salmonids to the low pressure associated with passing a turbine runner causes little direct mortality if the fish are acclimated to surface pressures. However, if the fish are acclimated to depth, injury rates increase due to expansion of gas in the swim bladders. Also, passage through areas of low pressure could produce injury and mortality under high saturation levels, and if fish respond to supersaturation by seeking greater depths. For new turbine designs, they recommend that increasing the nadir of the turbine pressure spike would reduce or eliminate fish injury and mortality caused by pressure.

Conclusions

Based on our review of the information discussed above, we conclude that:

- Estimates of survival through turbines vary highly with location (dam), method used, season, time of day, and salmonid species and life history type evaluated, making it difficult to provide summary statements useful to system managers.

- When we compare direct (balloon) estimates to direct and indirect estimates (PIT and radiotelemetry) such as described for McNary Dam in 2002, a significant component of overall turbine mortality is related to passage through the tailrace. Also, some nighttime estimates of turbine passage survival can be higher than comparable daytime estimates, suggesting that a part of this indirect effect is predation in the tailrace that is a result of turbine passage.

- A statistical relationship between fish survival and Kaplan turbine unit efficiency for Snake and Columbia River dams does not exist.

- Recent estimates of overall survival range from the low 70s to the low 90s, which while of interest, provides little useful information that can be used to alter turbine designs and operations. Since 1997 the Corps has recognized this and has been implementing the Turbine Survival Program to gain additional rigor and information on the relationships between juvenile salmonid condition and survival and the physical environment they are exposed to during turbine passage. This program, along with the DOE Advanced Hydropower Turbine System Program, have developed a significant body of knowledge through use of both engineering and biological studies to explore the causal mechanisms of injury and mortality. A great significant amount of progress has been made toward this goal. However, the answers to these questions remain illusive due to the inherent variability associated with fish passage through Kaplan turbines on the Columbia River.

- We suggest that additional, rigorous fields tests will be needed using large sample sizes and perhaps multiple evaluation methods to develop data sets of sufficient precision, that when combined with data from engineering evaluations, can make significant progress toward the goal of understanding the causal mechanisms of turbine mortality and make operational and design improvements to turbines on the Columbia River to improve the survival of juvenile salmonids migrating downstream.

KEY UNCERTAINTIES ASSOCIATED WITH JUVENILE PASSAGE

In recent years, the regional process known as the Plan for Analyzing and Testing Hypotheses (PATH), and the Independent Scientific Advisory Board (ISAB) for the Northwest Power Planning Council and the NOAA Fisheries have reviewed numerous data sets associated with dam passage on the Columbia and Snake Rivers. These reviews identified a number of uncertainties associated with the current information base.

Performance Measures

The ISAB suggested that there is a need for a "common currency" of stock-specific performance when measuring the results of hydropower system improvements (Bisson et al. 1999). They pointed out that management goals for population size are set at the individual stock or spawning population levels, whereas passage measures at dams are evaluated across the entire outmigration for a species or life history type.

The performance data collected depends on the methods used. For example, fyke-net estimates of FGE are species-specific or life history specific, but hydroacoustic estimates of FGE generate a value for all targets observed, including non-salmonids. Further, fyke-net estimates are generated during 2-hour evening sampling periods from the main portion of the outmigration to capture sufficient numbers of fish to estimate performance. Hydroacoustic estimates are benign, and can be collected for 24 hours per day over the entire migration season. Each passage research methodology has strengths and weaknesses, and many are used together to form a more complete data set. Only the PIT tag can be used to assess stock-specific performance, but it too has limitations, usually the number of fish that can be marked and detected. Until new sampling technologies are developed and the abundances of individual populations increase, the ISAB's comment on performance measures remains valid.

Selective Forces

The ISAB asserts that dams have acted as selective forces and reduced biodiversity (Bisson et al. 1999). They believe there is ample evidence to support this statement, based on observed variance in juvenile bypass system collection efficiencies for species and life histories. However, no direct measures of biodiversity are presented or cited to support their statement. The long-term effects that juvenile passage routes such as spill and surface bypass systems may have on all salmonid stocks and non-salmonid anadromous species are unknown. Therefore, uncertainty exists as to whether dam passage systems act as a selective pressure that reduces biodiversity within populations of anadromous salmonids.

Extra Mortality

Snake River spring/summer chinook salmon abundance declined precipitously after completion of the FCRPS (Raymond 1979, Schaller et al. 1999). The initial decline occurred in the early 1970s as Lower Granite, Little Goose, Lower Monumental, and John Day Dams were added to the existing FCRPS. The decline was roughly proportional to the direct mortality suffered by smolts during downstream migration through the completed system. Direct smolt mortality has decreased considerably over the past 2 decades (Williams et al. 2001) coincidental with installation of structural improvements at dams and initiation of operational procedures designed to enhance survival (Williams and Matthews 1995). However, despite the substantial gains realized in direct smolt survival, adult return rates of Snake River spring/summer chinook salmon have not increased to levels that were estimated to have existed prior to dam construction (Raymond 1988).

One of the most important and enigmatic questions currently facing the region is whether or not migration through the FCRPS, as currently configured and operated, causes mortality to anadromous smolts that is not expressed until after they have passed through the system. This hydropower-related delayed mortality has been termed “extra mortality” and was hypothesized during the PATH process to explain the change in productivity calculated for Snake River basin spring/summer chinook salmon populations compared to populations downstream of McNary Dam after construction of John Day, Lower Granite Little Goose, and Lower Monumental Dams (Schaller et al. 1996).

Evidence from spawner and recuit data indicated that productivity declined more for upriver stocks which were most affected by hydropower development, and that this reduction occurred primarily after completion of the three final dams on the Snake River (Schaller et al. 1999). Further, the differential decline was greater than could be explained by differences in direct mortality caused by the additional dams. Schaller et al. (1999) further argued that there was little evidence that factors unrelated to the FCRPS could account for the differences in productivity and survival between upstream and downstream stocks. On the other hand, Zabel and Williams (2000) and Hinrichsen

(2001) questioned this conclusion and provided evidence that several other factors could be responsible for the observed difference in productivity between salmon populations originating in the two areas.

Mechanisms that could potentially result in post-hydropower-system-passage delayed or extra mortality were postulated by Budy et al. (2002), but none have been confirmed with empirically-derived data to actually cause extra mortality. Hypotheses of how the hydropower system could produce extra mortality include the effect hydro-regulation has on flow and ocean entry timing, the cumulative effect of stress/injury associated with passing through consecutive turbines, bypass systems, and spillways, and the effects of stress, disease transmission, and delay on fish as they pass through bypass systems or fish ladders.

Alternatively, PATH also developed two other hypotheses to explain potential sources of differences in productivity between upstream and downstream areas (Marmorek and Peters 1998). The ocean regime shift hypothesis attributes the recent low survival of salmonids to cyclical changes in ocean conditions, and the stock viability degradation hypothesis represents the potential negative effects of many factors, including the effects of hatcheries on wild stocks, effects of diseases, bird predation associated with man-made dredge disposal islands in the estuary, and inbreeding depression.

Clearly, uncertainty exists over whether extra mortality, if it exist, is caused by the hydropower system or other factors. Analyses of hydropower system effects are confounded by changes in ocean productivity, Columbia River hydrology due to increased storage capacity, reliance on hatcheries to meet production goals, and other factors. The experimental data required to resolve this issue are not yet available. However, a new NOAA Fisheries study designed specifically to test the extra mortality hypothesis is scheduled to begin in spring 2004, with final results available by 2010.

Lamprey Passage

Pacific lamprey (*Lampetra tridentata*) ammocoetes burrow into sediment as filter-feeding benthic organisms. When large enough they metamorphose to the parasitic phase during July through October, and move downstream to the ocean between late fall and spring. Time in freshwater as larvae varies typically from 4-6 years, and some populations stay in freshwater up to ten months after metamorphosis. Lamprey rely on currents to be carried downstream, tailfirst, and the downstream migration occurs at night. Hydropower system effects on juvenile Pacific lamprey spatial distribution, how juveniles approach dams, timing of passage (interannual variability, seasonal variability, diel passage timing), and survival rates through reservoirs and past dams are unknown.

During FGE studies, NOAA Fisheries researchers have consistently observed juvenile lamprey distributed near the bottom of turbine intakes. Results from FGE studies at Bonneville Dam in 1998 are typical. Monk et al. (1998) captured a total of 308 juvenile lamprey in turbine intake fyke nets from April 25 through May 21. No fish were captured in the gatewell, therefore ESBS FGE was estimated to be zero. Of the 308 fish caught, approximately 70% were caught in the lower 4 nets, of nine nets total. Similarly, Long (1968) found juvenile lamprey were concentrated near the center and bottom of turbine intakes at The Dalles Dam in 1960 (fish guidance screens were not installed in the intakes).

A percentage of lamprey do migrate high in the water column and are intercepted by screens during ESBS FGE testing. Of these, some have become impinged on the ESBS when debris brush sweeps were not functioning properly. For example, at The Dalles Dam in 1993 the debris sweep on a prototype ESBS was disabled to mount video cameras on the screen to observe juvenile salmon behavior near the screen face. The screen was operated in this condition for a 7-day period, raised for inspection, and 50 to 100 dead lamprey juveniles were observed. Similarly, Monk et al. (1998) found that when a test ESBS at Bonneville Dam was raised for inspection, the debris sweep on one of the screens was not adjusted properly. The top foot of the screen was not being cleaned, and two juvenile lamprey were observed in this section.

ADULT PASSAGE

Background

Bjornn and Peery (1992) synthesized the available information on adult salmonid migrations through the lower Snake River. They provided a thorough compilation of published and unpublished information on adult steelhead and chinook salmon migrational behavior available through 1991. This review also included information on sockeye and coho salmon from sites outside the Snake River. We utilize the Bjornn and Peery (1992) synthesis, and add information collected from studies since 1991. We do not describe the habitat lost to adult migrants due to construction of the hydropower system. The scope of this document is to describe the available scientific information that pertains to passage through the hydropower system as it is currently configured. All eight mainstem dams from Bonneville through Lower Granite Dams provide upstream passage for adult salmon and steelhead through one or more fish ladders.

Adult Passage System Criteria and Issues

The ladder systems are operated according to criteria developed by the COE, NOAA Fisheries, and state and tribal fishery co-managers. Each criterion is based on results of biological testing to determine the hydraulic conditions that maximize fish attraction and minimize delay. The COE annually coordinates with salmon managers and publishes the Fish Passage Plan (FPP) (COE 1999d). The FPP provides detailed operating procedures and criteria for adult fish passage facilities and special operations to accommodate research. This includes criteria on water depth and head on the entrance gates, powerhouse collection channels, floating orifices, ladder flow, counting windows, and ladder exits.

There is a significant backlog of unfunded maintenance and repair projects on entrance gates and gate lifting machinery at Snake and Columbia River dams. Also, at some Snake River project entrance gates the auxiliary water supply (AWS) systems cannot supply sufficient attraction water to meet the required minimum of 1.0-ft head differential between the adult collection channel and tailrace. This situation can develop during low tailrace conditions, which occur when river flows are low and the downstream project is operated within the lower foot of the operating range of the reservoir.

At Lower Monumental, Little Goose, and Lower Granite Dams, all ladder entrances are served by a single AWS pump station. Ladder entrances located on the opposite side of the spillway from the pump station cannot satisfy the established flow criteria. During high river flow conditions, submergence and head differential criteria for south entrances at Lower Monumental and the north entrances at Little Goose and Lower Granite Dams are not met.

The 1995 BiOp (NMFS 1995a) requires the COE to provide an emergency source of water to satisfy fishway criteria if the main AWS system fails at each project. Engineering studies have been completed for all projects except Little Goose and Lower Granite Dams. The Walla Walla District has selected a contractor to complete the design studies for these projects. John Day and McNary Dams were able to meet the BiOp requirement with existing facilities. Engineering alternatives for meeting emergency AWS capability at The Dalles Dam and Bonneville Second Powerhouse are being evaluated.

Jumping in and from fish ladders is another adult passage issue. This behavior is typically associated with the transition from overflow weirs to vertical slot weirs, and with diffuser flow at the top of ladders (Dresser 1998). This occurs primarily with steelhead at both John Day Dam ladders. However, at Lower Granite Dam faster passage times and more direct passage routes were observed for both Chinook salmon and steelhead when weir panels were in the down position (Naughton and Peery 2003).

Migration Behavior

Migration behavior of adult salmonids in the Columbia and Snake River drainages has been documented using uniquely-coded radio transmitters. The transmitters are implanted in the fish's esophagus and transmitters are detected using an extensive array of fixe-site antennas. Matter and Sandford (2003) compared the upstream ravel times of PIT-tagged and radio-tagged spring/summer chinook salmon from Bonneville to Lower Granite Dams in 2000. There was no evidence that radio-tagging affected chinook passage rates. In fact, upstream travel time for adult Chinook salmon outfitted with PIT tags (15.9 d) was significantly longer than passage time of fish outfitted with radio transmitters (14.1 d).

Salmonids moving up the Columbia and Snake Rivers and through reservoirs tend to stay near the shore. This seems to hold true for all species. This trend is documented by tracking radio-tagged fish, and data from individual fish tracks indicate crossing between sides of the river or reservoir occurs at what appear to be random locations. Natural migration behavior is a concern when designing fishway entrances and exits.

Keefer et al. (2003a) showed that locating large fishway entrances on shorelines produces high net entrance rates (number of fish entering orifices minus number of fish leaving), and high fishway entrance efficiency (entrances that produce passages).

Locating fishway exits on shorelines increases the probability that fish will continue upstream after exiting the fish ladders and reduces the chance of falling back downstream through the spillway. For example, the south fish ladder at Bonneville Dam exits into the forebay at Bradford Island, which affects the rate of fallback. Adults that move upstream from the exit arrive at the spillway, and adults that move downstream arrive at the powerhouse. When only fallbacks within 24 hours of passage are considered, 92 to 97% of all fish that fell back at Bonneville Dam between 1996 and 1998 had used the Bradford Island fishway (Bjornn et al. 1998a).

Where fish first approach a dam changes as the proportion of river discharge changes from powerhouse to spillway. With no spill most fish approach the dam on the shoreline adjacent to the powerhouse, and as spill starts a portion of those fish move to the shoreline adjacent to the spillway. At moderate spillway rates, more fish approach the dam at the junction of the powerhouse and the spillway. Under rare high flow and high spill events, when turbidity is also usually high, upstream movement of adults can slow considerably for a few days until the event subsides.

Once at the dam, fish search across the downstream face looking for passage routes. Entrance preferences are for deep/wide openings with significant attraction flow. Shallower/smaller openings tend to have low numbers of entrances and negative net entrance rates. Median times for chinook salmon to first enter fishways were 1.9 to 2.6 hours at Ice Harbor and Lower Granite Dams in 1993 and 1994 (98.7 and 98.9% entered in less than 5 days). At Lower Monumental and Little Goose Dams, 98.5 and 98.2% entered in less than 5 days, and median times to first entries were 4.6 and 3.9 hours, respectively (Bjornn 1998b).

Preliminary data on first entry time from ongoing studies in the lower Columbia River are comparable to those at the three lower Snake River dams. At Bonneville Dam, chinook salmon median first entry times were 2.0 and 2.2 hours in 1996 and 1997; steelhead median first entry times were 1.9 and 0.3 hours in 1996 and 1997, respectively. First entry times ranged from 0 (when the first at-dam record was in the collection channel) to 20 days for chinook salmon and 17.8 days for steelhead (Lowell Stuehrenberg, NOAA Fisheries, Pers. commun., March 2000).

Bjornn et al. (1998d) concluded that when powerhouses are not at full load, changing the end of the powerhouse where turbines were operating had little, if any, influence on the time for steelhead to approach, enter, or pass fishways at Snake River dams. They saw a slight change in first approach sites, but not first entrance locations. Bjornn et al. (1998e) did not detect any differences in chinook and sockeye salmon or steelhead entrance locations or passage times when turbine Unit 1 (and its associated discharge located near the south shore adult entrance) at John Day Dam was operated at 150 and 100 MW.

Fishway fences installed adjacent to the north powerhouse entrances at Little Goose and Lower Granite Dams in 1991 to reduce fallouts as fish moved upstream in the collection channel were not effective (Bjornn et al. 1995). Funneling the down-channel moving fish away from the entrances at the downstream end of the collection channels at Little Goose Dam in 1994 improved the net entrance rate at those entrances (Bjornn et al. 1998b).

A high rejection rate of the transition area between the collection channels and the fish ladders has been documented for spring/summer chinook salmon and steelhead (Stuehrenberg et al. 1995, Bjornn et al. 1998b). For steelhead the rejection rates range from 46 to 71% on their first approach to the transition area at Snake River dams. Similar rates were found for Chinook salmon at both Columbia and Snake River dams in 1996 (Keefer et al. 2003a).

In 1994, the University of Idaho Cooperative Fish and Wildlife Research Unit installed radiotelemetry antennas in the transition pools of the four lower Snake River projects and tracked the progress of 220 to 246 steelhead at each project. From 36 to 61% of the fish turned around in the transition pools, moved downstream, and exited the fishway at least once (Bjornn et al. 1998f). An additional 8 to 27% moved downstream in the fishway but did not exit.

Between 4 and 35% of spring/summer chinook salmon turned around in the transition pool and headed downstream at McNary Dam and Ice Harbor Dam. At Bonneville Dam and Lower Granite Dam, 53 to 55% turned around in the transition pool (Keefer et al. in review (a)). Similarly, fall Chinook entering fishways at the lower Columbia River dams turn downstream in the transition pool more so than any other fishway segment (Brian Burke, NOAA Fisheries, Pers. commun., October 2003).

Several hypotheses have been offered to explain this transition and junction pool behavior: a) velocities are too low and unsteady, b) inadequate flow rates and velocity to attract fish to the submerged section of the ladders and through the orifices at the base of the submerged ladder weirs, c) seasonal and intermittent temperature gradients between the ladder flow and the diffuser flow, d) high flow rates through large floor diffuser areas obscure attraction flow at the base of the ladders, and e) fish may be wary and reluctant to move into confined ladder pools.

The COE (COE 1994) measured velocities in the collection channel and transition pools at Little Goose and Lower Granite Dams. The velocities were lower than the minimum criterion of 1.5 fps, non-uniform and unsteady. The COE tested the first two hypotheses (above) by installing baffles in the transition pool at Little Goose Dam in 2000 and monitoring fish fallback using radiotelemetry. Results are pending.

Radiotelemetry studies in 1993 and 1994 have shown that adult fish both enter and exit the floating orifices in the powerhouse collection channels of Snake River dams. The net entry rate indicated that for fish that entered the collection channel via the orifices, more fish left via the orifices than stayed in the collection channel (Bjornn et al. 1998b). The COE's Fish Passage Operations and Maintenance (FPOM) committee concluded that closing the floating orifices at Snake River dams would improve the operation of the adult fishways because more water would be available for the main fishway entrances and maintenance would be improved because the bulkhead could be sealed and the collection channels dewatered on a more frequent basis.

The earliest study of migration rates in the Snake River prior to impoundment was conducted from 1954 to 1957 (Oregon Fish Commission 1960). Chinook salmon migration rates averaged 17.7 to 24.1 km/day. Steelhead migration rates varied with the season; spring rates averaged 11.3 to 16.0 km/day, whereas fall rates averaged 8.0 to 9.7 km/day. Sockeye averaged 19.3 km/day to a weir at Redfish Lake, more than 600 km upstream.

In a separate study, steelhead tagged at McNary Dam in January, February, and April migrated at an average rate of 3.2, 3.9, and 12.2 km/day, respectively. The effect of season and water temperature on steelhead migration rates was also seen in studies conducted from 1969 to 1971, where steelhead had summer migration rates of 10.7 to 16.7 km/day, and late fall migration rates as low as 0.5 km/day. Bjornn and Peery (1992) concluded that in the Snake River prior to impoundment, salmon migrated from 18 to 24 km/day, and steelhead migration rates varied with season and water temperature and ranged from 11 to 17 km/day during the spring and summer to as low as 0.5 km/day during the late fall.

Spring/summer chinook salmon migration rates through the Snake River reservoirs in 1991 to 1993 ranged from 31 to 65 km/day while migration rates through free flowing river sections above Lower Granite Dam ranged from 10 to 30 km/day (Bjornn 1998c). Travel through the Columbia River hydrosystem is slower, with median rates of 13 to 33 km/day for spring chinook in 1996-2000. Migration times from Bonneville to McNary ranged from 6 to 20 days for individual fish. For all groups, migration rate variation between years was correlated to mean flow and temperature; the date of migration was the most influential predictor of migration rates, followed by river discharge (Keefer et al. in review (a)) .

The median steelhead migration rate through the Snake River reservoirs in 1993 was 30 km/day while migration rates through free flowing river sections were generally less than 11 km/day (Bjornn 1998c). Based on the passage times at the dam and faster passage in the reservoirs than in free flowing rivers, Bjornn et al. (1999) estimated that median time for salmon to pass the four dams and reservoirs in the lower Snake River in 1993 was the same or less with dams as without the dams.

In addition to the radiotelemetry data presented above, in 1998, 38 hatchery steelhead of known Snake River origin (based on PIT tags) were detected at both Bonneville and Lower Granite Dams (through December 31, 1998). Their median time for passage between the two dams was 32 days, or 14.4 km/day (ranged from to 5.7 to 40.7 km/day for the 460 km reach). Also in 1998, 22 spring/summer chinook salmon of known Snake River origin (based on PIT tags) were detected at both Bonneville and Lower Granite Dams. The median time for passage between the two dams (a total of 460 km) was 16.4 days, or 28.0 km/day (ranged from to 12.4 to 47.9 km/day).

Also in 1998, 38 fall chinook salmon of known Snake River origin (based on PIT tags) were detected at both Bonneville and Lower Granite Dams. Of these fish (1 wild and 37 hatchery fish), the median time for passage between the two dams (a total of 460 km) was 12.0 days, or 38.3 km/day (ranged from to 13.9 to 51.1 km/day) (Lowell Stuehrenberg, NOAA Fisheries, Pers. commun., September 1999). This suggests that the 1998 migration rate of PIT-tagged steelhead through the FCRPS falls within the range observed in the Snake River prior to construction of dams, and the migration rate of PIT-tagged spring/summer chinook salmon slightly exceeds the range reported by Bjornn and Peery (1992) for chinook salmon (run-type was not specified).

NOAA Fisheries compared the upstream travel times of PIT-tagged and radio-tagged spring/summer chinook salmon from Bonneville to Lower Granite Dams in 1998. There was no significant difference in upstream travel time between adult chinook salmon outfitted with PIT tags (N = 22) and radio transmitters (N = 216) when seasonal passage trends were taken into account. The median travel time was 16.4 d for PIT-tagged fish (min 9.6 d; max 37.0 d), and 18.4 d for radio-tagged fish (min 8.0 d; max 85.7 d) (Alicia Matter, NOAA Fisheries, Pers. commun., March 2000).

Raymond (1964) compared the median migration timing of sockeye and chinook salmon past Bonneville and Rock Island Dams between 1938 and 1950 when no other dams existed in the hydropower system corridor. The mean difference in passage time between Bonneville and Rock Island Dams of the annual median sockeye salmon passage at each dam was 16.5 days (range 7 to 27 days). NOAA Fisheries computed the same statistic for the period between 1985 and 1999 and found a mean difference in passage time of 15 days (range 11 to 19 days). Quinn et al. (1997) also studied migration rates of Columbia River sockeye salmon. They found that travel time has decreased in the last 40 years between Bonneville and McNary Dams, but was unchanged between McNary and Rock Island Dams. They also found that river temperatures between McNary and Rock Island Dams actually decreased between 1933 and 1993 and speculated that the reduction in temperatures and reduced water velocities may have resulted in energetic savings.

A substantial percentage of adult salmon and steelhead passing dams have been observed to fall back through spillways and turbines at certain dams under certain conditions (Bjornn and Peery 1992). High fallback rates are usually associated with high river flows and spill, as well as the location of fishway exits relative to the spillways (Boggs et al. unpub. data). Liscom et al. (1979) concluded from several fallback studies that fallback rates can be high at times, but few fish are injured or die as a direct result of fallback. Migration times are increased if the fish must reascend the dam.

Bjornn and Peery (1992) present fallback rates from over 50 radiotelemetry studies of chinook and sockeye salmon and steelhead at various dams from 1966 through 1985. Keefer and Bjornn (1999) estimate, based on radiotelemetry, that ladder counts at Bonneville Dam are overcounted by 13.5 to 19.3% for spring/summer chinook salmon from 1996 through 1998, by 4.7 to 8.2% for steelhead from 1996 through 1997, and by 12.6% for sockeye in 1997 when fallback and reascension are taken into account. Similarly, from 1996 to 2001, passage overestimation over the lower Columbia and Snake River Dams ranged from 1 to 16% for spring/summer Chinook, 1 to 38% for fall Chinook, and 1-12% for steelhead (Boggs et al. unpub. data).

Between 1996 and 2001, fallback rates for spring/summer chinook salmon in the lower Columbia River ranged from 2 to 20% (1 to 14% at the Snake River Dams, Boggs et al. unpub. data). Fallback rates for steelhead during that same time period ranged from

5 to 13% at the lower Columbia River Dams and from 2 to 9% at the Snake River Dams. From 1998 to 2001, fallback rates for fall Chinook ranged from 2 to 12% at the lower Columbia River dams and from 3 to 58% at the Snake River Dams.

The Bradford Island ladder exit at Bonneville Dam has been associated with the highest fallback rates on the Columbia River. Bjornn et al. (1998a) found that fallback rates were 2.5 to 3.7 times higher for spring/summer chinook salmon that passed via the Bradford Island ladder than the Washington-shore ladder, and 16 to 20 times higher for sockeye salmon. In 1996, almost all steelhead that fell back passed via the Bradford Island ladder.

In addition to fish that fall back within 24 hours of exiting the fish ladder, a large number of fallbacks occur after fish have migrated significant distances upstream. Dam operating procedures or environmental conditions at individual dams are unlikely to affect fallback associated with this behavior. The cause of this wandering behavior has not been determined. Potential causes include searching for their natal tributary, natural survival adaptation, the use of hatchery broodstock not native to the drainage, and some level of homing impairment from having been transported. Mendel and Milks (1996) attributed much of the fall chinook salmon fallback they observed to poor entrance conditions at the Lyons Ferry Hatchery and extensive wandering up and down the Snake River and into tributaries. They estimated that the fallback at Lower Granite Dam was 16 to 39% in 1993, and 30 to 41% in 1992. They noted that over 80% of the radio-tagged fall chinook salmon that entered Lyons Ferry Hatchery each year had reached Little Goose Dam before descending to the hatchery.

Survival

Losses can be broken down into two segments 1) losses between dams, defined as the difference between fish counted at consecutive dams minus those accounted for between those dams (harvest and tributary turn off), and 2) losses between the upper dam and the spawning grounds or hatchery.

Losses between dams that are based on dam counts are susceptible to counting errors and assumptions. For example, the largest portion of the over-counts at the dams is likely due to multiple counts of fallback fish. If these fallbacks occur at similar rates at two dams, the count error between those dams based on fallback is not affected. If the lower dam has a higher fallback rate than the upper dam, losses between the two dams would be biased. Radiotelemetry that allows individual fish to be tracked provides a more accurate assessment of actual passage rates and behavior than dam counts, although using radiotelemetry results assumes that tagged fish represent the population at large and that tagging and release do not affect the behavior of the fish. Further, estimates of survival based on radiotelemetry may be conservative because they include both actual mortality and unaccounted-for losses, such as tag regurgitation, unreported harvest, and tributary turnoff that is missed or not monitored. PIT tag survival estimates between detection sites are also accurate, though the relative scarcity of detectors relative to radiotelemetry necessitates more genera and limited survival conclusions.

With respect to losses between the upper dam and the spawning grounds or hatchery, little early information is available. Survival to spawning for spring/summer chinook salmon was estimated at 55% for fish counted at Ice Harbor Dam during 1962-1968 when only one dam was present, and 46% for the 1975-1988 when all four dams were in place (Bjornn et al. 1998c). These were developed by relating counts at Ice Harbor Dam with redd counts in Snake River tributaries.

More recently, with all dams in place, radiotelemetry has been used to estimate the survival of spring/summer chinook salmon from Ice Harbor Dam to spawning grounds or hatcheries. Bjornn et al. (1995) estimated survival through this reach was 77% in 1993, 73% in 1992, and 54% in 1991 for fish that were radio tagged at John Day Dam in 1993, and Ice Harbor Dam in 1992 and 1991. Caution should be used in comparing survival between the pre-development and post-development periods since the two techniques discussed above are very different.

For spring/summer chinook salmon radio-tagged at Ice Harbor Dam, survival from Ice Harbor Dam to Lower Granite Dam was 80.8 and 74.4% in 1991 and 1992, respectively. For spring/summer chinook salmon radio-tagged at John Day Dam in 1993 survival from Ice Harbor Dam to Lower Granite Dam was 85.9% (Bjornn et al. 1999). For spring/summer chinook radio-tagged at Bonneville Dam in 1996, 1997, and 1998, survival from Ice Harbor Dam to Lower Granite Dam was estimated two ways. First, for only those fish that passed the top of Ice Harbor Dam and moved upstream. Second, for all fish that approached Ice Harbor Dam (whether or not they passed the dam). Survival from Ice Harbor Dam to Lower Granite Dam for fish that passed the top of Ice Harbor Dam was 96.4, 97.7, and 98.4% in 1996, 1997, and 1998, respectively. Survival from Ice Harbor Dam to Lower Granite Dam for fish that approached Ice Harbor Dam was 92.8, 95.2, and 98.4% in 1996, 1997, and 1998 respectively (in 1998 both survival estimates were 98.4% because all fish that approached Ice Harbor either passed the top of the dam, or were harvested or detected downstream) (Lowell Stuehrenberg, NOAA Fisheries, Pers. commun., February 2000).

Lower Snake River reach survival estimates are based on tagging conducted at Ice Harbor, John Day, and Bonneville Dams. Bjornn et al. (1999) estimated survival through the lower Snake River at 80.8 and 74.4% for 1991 and 1992, respectively (fish were marked at Ice Harbor and released at Hood Park downstream of the dam). Survival of fish from Ice Harbor Dam to Lower Granite Dam was 85.9% in 1993 for fish marked at John Day Dam, and 92.9% in 1996 for fish marked at Bonneville Dam.

Estimates based on tagging conducted at John Day and Bonneville Dams probably do not include any effects from trapping, tagging, and release since any effects associated with radio tagging likely occurred prior to entering the lower Snake River. Bjornn et al. (1998c) found that the survival of spring/summer chinook salmon from Ice Harbor Dam to over Lower Granite Dam from 1991 to 1993 was similar between fish marked with radio transmitters (83 to 87%) and unmarked fish (70 to 92%), based on comparing radiotelemetry data with dam counts. This suggests that there is no effect of trapping fish at Ice Harbor Dam on survival estimates.

Recent outfitting of Bonneville Dam with a complete PIT tag detection system (winter 2000-2001) combined with existing detection capabilities upriver has allowed for survival analysis for adults migrating from Bonneville Dam to Lower Granite Dam. Data from transportation studies conducted by NOAA Fisheries (results from Doug Marsh, NOAA Fisheries, Pers. commun., October 2003) shows that for spring/summer Chinook salmon adults, survival varied between 80 and 85%, with no effects of transportation as juveniles. Steelhead survival for this section was 70% for adults that were transported as juveniles, and 80% for non-transported fish.

Assigning mortality associated with fallback to dam operations or behavior is difficult because some fish may have “over-shot” and then return to lower river tributaries. Bjornn (1998a) observed fallback mortality of 8% for sockeye salmon at Bonneville Dam (a species with no spawning below Bonneville Dam). Mendel and Milks (1996) estimated fall chinook fallback mortality at 26 and 14% in 1993 and 1994, respectively, for fish that fell back through one or more of the four lower Snake River dams. This higher mortality for fall chinook occurred during periods of no spill, when the fallback was assumed to have been through turbines. Keefer and Bjornn (1999) used radio-tagged fish known to have passed Bonneville Dam in 1996 to estimate survival to tributaries or the top of Priest Rapids Dam. Steelhead and spring/summer chinook salmon that did not fall back over Bonneville Dam had survival rates that were 3.8 to 5.2% higher than fish that did fall back. Fish that did not fall back over any dam had survival rates were 3.0 to 5.4% higher than fish that did fall back.

Keefer and Bjornn (1999) reported that for all 7 dams studied in 1996, the median passage time to pass a dam were higher for spring/summer chinook salmon that fell back at a dam one or more times. The potential impacts of increased passage times due to fallback on adult delayed mortality and reproductive success are unknown.

Zero Flow Operations

Reducing powerhouse output to near zero river discharge during hours of darkness was conducted to preserve water for periods of higher power demand. Bjornn et al. (1998g) found no clear evidence that reducing flows to near zero at night affected adult steelhead migration rate, proportion of fish passing dams, proportion of fish captured in the fishery, or proportion of fish returning to hatcheries in 1991 and 1993. A consistent pattern of slower steelhead migration in the early and late portions of the runs was found, which was associated with warm and cold water temperatures, respectively. Adult fishways were operated continuously throughout the tests.

Water Temperature

Bell (1991) describes the preferred temperature range and upper and lower lethal temperature limits for most salmonid species. For example, chinook salmon prefer 7 to 14.5ºC (45 to 58ºF) water, and have a lower and upper lethal limit of 0 and 25ºC (32 and 77ºF), respectively. Summer water temperatures in the Snake and Columbia Rivers often exceed 21ºC (70°F), and water temperatures in ladders can be even higher than ambient river temperatures.

River water temperatures greater than 20°C cause travel times between dams to increase for migrating salmonids (Peery et al. 2003). Affected fish may abort their migrations or seek cooler water that may not be in the direct migration route to their spawning site. Snake River fall chinook salmon and steelhead often slow their migration through the Columbia River and delay entering the Snake River when water temperatures are high (Stuehrenberg et al. 1978). Instead, these fish seek temporary refuges in lower Columbia River tributaries en route to their final destinations at up-river locations.

Of 125 Chinook recorded at Ice Harbor Dam in 1996, 20% had been recorded in lower Columbia River tributaries (Bjornn et al. 2000). For steelhead, 65% of fish passing Lower Granite Dam were observed in one or more lower tributaries (Keefer et al. 2002). In both cases, peak tributary use coincided with peak mean daily temperatures at dams (approximately 22°C), while recorded temperatures in the tributaries were less than 20°C. Delays associated with high water temperatures can subsequently affect reproductive success. Of 71 radio-tagged sockeye salmon that held in the Columbia River until the Okanogan River cooled in 1992, only 24 (33.8%) survived to the spawning grounds (Swan 1994). Delay caused by high water temperatures could also impact the reproductive success of fall chinook salmon which spawn in the fall, as discussed below. The effect of delays on steelhead, which spawn in the spring, are unknown.

The 1995 BiOp (NMFS 1995a) requires the COE to provide water temperature control in fish ladders. The University of Idaho Cooperative Fish and Wildlife Research Unit began monitoring temperatures in the forebays and adult fishways at Ice Harbor and Lower Granite Dams in 1995. Temperature changes along the length of the fish ladder were found to be slight: less than 0.5°C with differential increases to 2°C found on a few occasions (Keniry and Bjornn 1998). Temperature differences upstream and downstream from diffusers were also minimal.

However, the use of warm surface water from reservoirs affects temperature in the entire fishway, and a discontinuity may occur at the transition pools at the base of fishways. Here, cool water is added to promote flow at fishway entrances. At Lower Granite Dam, this system has resulted in temperatures up to 4°C warmer in transition pools than at fishway entrances (Peery et al. 2003). The COE also monitors water temperatures in the ladders at John Day Dam, which exhibits similarly steep thermal gradients near the bottom of its south ladder. Salmonid passage times at John Day Dam are longer as a result, and more fish exited fishways as ladder temperatures increased, which further contributed to extended passage times (Keefer et al. 2003b).

Since 1992, Dworshak Dam has been operated to provide cold water for lower Snake River temperature control to benefit outmigrating juvenile fall chinook salmon. Up to 1.2 million acre feet of water at temperatures from 5.6 to 13.3°C (typically 8.9 to 11.1°C) have been used to augment flows and reduce high water temperatures. These releases are managed by regional salmon and reservoir managers to improve migration conditions for migrating smolts, and are typically made during July and August. Karr et al. (1998) estimated that because of water releases from Dworshak Dam in 1994-1996, water temperatures were 3.1 to 5.3ºC cooler for 18 to 32 days at Lower Granite Dam, and 2.4 to 3.1ºC cooler for 14 to 43 days at Ice Harbor Dam. However, this estimated cooling is within the range of normal temperature variation at Ice Harbor Dam (Ted Bjornn, Idaho Cooperative Fish and Wildlife Research Unit, Pers. commun., January 2000).

Dauble and Mueller (1993) suggest reducing mainstem water temperatures to below 21°C could reduce risk to populations of migrating adult salmon. It appears that operation of the Hells Canyon Complex has decreased late winter/spring temperatures and increased September/early October temperatures in the Snake River (Richie Graves, NOAA Fisheries, Pers. commun., January 2000). Ted Bjornn (Idaho Cooperative Fish and Wildlife Research Unit, Pers. commun., January 2000) found a similar trend when he plotted mean water temperatures at Ice Harbor Dam from August through October, and compared two periods:1962 to 1968 and 1975 to 1989. The 1975-89 period was cooler during August but warmer in September and October than the 1962-68 period.

The effect of temperature on adult spawning may also affect Snake River fall chinook salmon fry emergence timing. NOAA Fisheries estimates that emergence has been delayed by about 2 weeks in higher flow years and up to nearly 4 weeks in lower flow years. This was developed by comparing recent juvenile run-timing information to trapping studies of juvenile fall chinook emigrating into Brownlee Reservoir in the late-50s and early-60s (Richie Graves, NOAA Fisheries, Pers. commun., January 2000).

Recently, comparisons have been made of the migration timing of steelhead and fall chinook salmon from Bonneville Dam to Ice Harbor Dam, based on radiotelemetry studies conducted from 1996 to 1998 and dam counts from 1991 to 1998. Plots were made comparing date, cumulative proportional count at Ice Harbor Dam, and river temperature (turbine scroll case). The plots indicated several general trends. First, steelhead passage at Ice Harbor Dam is slightly later than fall chinook salmon. Second, a high proportion of fall chinook salmon move into the Snake River when water temperatures exceed 20ºC.

This was particularly apparent in 1998 when over 80% of the fall chinook salmon crossed Ice Harbor Dam before water temperatures dropped below 20ºC. Third, based on fall chinook salmon dam counts (migration peaks and dates) from Bonneville Dam to Ice Harbor Dam for 1991 through 1998, fish appeared to move rapidly upstream through the system relative to spring/summer chinook (also noted in Matter and Sandford 2003). If fall chinook salmon migrations over Ice Harbor Dam are later than normal, it appears their migration dates are delayed from Bonneville Dam upstream, rather than being delayed by specific warm water conditions at the mouth of the Snake River (Ted Bjornn, Idaho Cooperative Fish and Wildlife Research Unit, Pers. commun., January 2000). Furthermore, overall return migration timing is governed by both river environment (discharge levels) and stock adaptations (genetic influence on migration timing) for spatially separated Columbia and Snake River salmonid stocks (Keefer et al. in review (b)).

Dissolved Gas Supersaturation

Mortality of adult salmon from gas bubble disease (GBD) has occurred in the Columbia River intermittently since the first dams were constructed (see Dissolved Gas Supersaturation discussion, above). However, effects of GBD on adult salmonids are less understood than are the effects on juveniles. Relationships between TDGS exposure, GBD signs, and mortality are not defined. Depth distributions of upstream migrants are poorly documented, thus, the mitigative effects of hydrostatic compensation at ambient TDGS is unknown. Also, there are no research data sufficient for determining a threshold TDGS level that ensues successful spawning of upstream migrant adult salmonids (NMFS 1997).

Following implementation of voluntary spill to enhance dam passage for juvenile salmonids, monitoring was initiated to examine upstream migrating adult salmonids for GBD signs. Beginning in 1994, adult fish were examined annually at Bonneville and Lower Granite Dams and intermittently at Ice Harbor and Priest Rapids Dams. Under voluntary spill conditions where reservoir and tailrace TDGS is limited to 115 and 120%, respectively, signs of GBD were not seen and effects were assumed to be benign. Monitoring was also done at Three Mile Dam on the Umatilla River, OR. However, facilities to capture and examine adult salmonids at this site are limited. Thus, the ability to obtain representative samples from locations where the greatest impacts may occur, for example downstream from McNary Dam, is limited. Also, spawning success in relation to TDGS has not been monitored.

The only period in the 1990s when GBD signs were readily observed on adult salmonids was following the spring freshet in 1997 (NMFS 1998a). However, GBD signs were documented only at a few sites and no estimates of impact could be made. During that period, TDGS downstream from Bonneville Dam exceeded 135% for 16 days and 130% for 24 days. TDGS exceeded 125% for extended periods at other river reaches.

At Bonneville Dam Second Powerhouse, daily prevalence of GBD was high for sockeye salmon (14 to 100% for more than 3 weeks) and steelhead (6 to 50% for 2 weeks). Chinook salmon during the same period showed relatively few signs, with 0 to 6.5% prevalence. No samples were collected from fish traversing the Bonneville Dam spillway tailrace, where TDGS was highest. No GBD signs were observed at Lower Granite Dam; however, fish were not examined until they had spent several hours in the low TDGS conditions of the fish ladder; GBD signs may have disappeared during ladder passage. At Priest Rapids Dam, sampling took place after TDGS decreased to moderate levels of 113 to 124%. In 1996, prior to spillway deflectors being installed, average

TDGS downstream from Ice Harbor Dam was very high for almost one month, generally exceeding 135%. Adults examined at Lower Granite Dam showed no signs of GBD. However, this site is 172 km upstream from Ice Harbor Dam. Further, TDGS in the Little Goose reservoir (59 km long) was less than 125% for all but 3 days.

Head burn (exfoliation of the skin and underlying connective tissue) on the top or sides of the heads of adult salmonids was observed in the Snake River during periods of high spill and was commonly thought to be a sign of GBD. Head burn was considered by the Gas Bubble Disease expert panel held by NOAA Fisheries in 1995, but given a low research priority ranking (NMFS 1996). Efforts to address the cause of head burn were transferred to the COE Fish Passage Operations and Maintenance Coordination Team who have discussed the potential causes of head burn. Elston (1996) conducted clinical evaluations of fish with typical head burns from Lower Granite Dam and suggested that head burns were caused by mechanical abrasion and laceration, rather than necrosis associated with subcutaneous emphysema from GBD.

Monitoring for head burn has been conducted at Lower Granite Dam by NOAA Fisheries since the early 1990s. From 1993 through 1999 the percentage of adult chinook salmon with head burn ranged from 0 to 9.8%. Monitoring for head burn has been conducted at Bonneville Dam by the Columbia River Inter-tribal Fish Commission since the mid-1990s. From 1997 through 1999 the percentage of adult chinook salmon with head burn was <1% (Larry Basham, Fish Passage Center, Pers. commun, February 2000). The exact cause of head burn remains unknown. Monitoring will continue to document the rate of injury. Research on the migration behavior of fish with head burn will be needed to relate the injury to survival. For example, Bjornn et al. (1995) found in 1993 that of 66 radio-tagged chinook salmon that were noted as having head scrapes or injuries, 38% did not migrate to known spawning areas and were classified as possible prespawning moralities.

Kelts

Post-spawning, downstream migrations of the iteroparous component of Columbia Basin steelhead (*O. mykiss*) have been documented throughout the existence of the FCRPS (Long and Griffin 1937, Whitt 1954, Evans and Beaty 2000, 2001, Wertheimer et al. 2001, 2002) and the potentially negative effects of the hydropower system on iteroparity rates recognized (NPPC 1986, ISG 1996). Recent information for upper Snake River Basin areas indicates that repeat spawning rates in the Snake River probably average less than 2% (Evans and Beaty 2000). However, the specific effects of the hydropower system on the survival and reproductive success of Columbia Basin kelts are poorly understood . Only recently have studies been directed at conclusively identifying kelts (vs. pre-spawners or fallbacks) and documenting their downstream timing and project-specific passage.

In 2000, the COE initiated a program to study the downstream passage behavior of kelts at Bonneville Dam. Overall, 39.5% of radio-tagged kelts passed through the spillway, 39.5% through Powerhouse I, and 21% through Powerhouse II. At Powerhouse I, 83% (43/52) kelts passed through turbines units, 8% (4/52) passed through the prototype surface collector into the ice and trash sluiceway , and 10% were guided by traveling screens into gatewells and the juvenile bypass system. Median forebay residence times were longest at Powerhouse I (8 hours, 39 minutes), followed by Powerhouse II (4 hours, 34 minutes), and were significantly shorter at the spillway (15 minutes).

In 2001, kelt sluiceway passage efficiencies of 69% and 89% were estimated at The Dalles and Bonneville Dams, respectively (Wertheimer et al. 2002). No benefits were observed from blocked trashracks at The Dalles Dam. As in 2000, turbine intake guidance screens were inefficient in guiding kelts away from turbines. Spilling water significantly reduced the travel and passage times of kelts through the projects and pools in the lower Columbia River. At The Dalles Dam, 30% spill achieved a kelt passage efficiency of 99% and sluiceway effectiveness increased (Table 11).

Studies in 2002 focused on evaluating in-river abundance of kelts at McNary and John Day Dams (Wertheimer et al. 2003). An estimated 14,057 kelts were present in McNary reservoir and an estimated 13, 081 kelts passed John Day Dam during a 9-week period. Passage through John Day Dam under 24-hour spill (30%) and nighttime spill (0% day / 54% night) were evaluated (Table11) and sluiceway effectiveness was high at The Dalles and Bonneville Dams. In addition, to evaluate possible effects of lower Columbia River dams and reservoirs on respawning rates, kelts were PIT tagged at John Day Dam and returned directly to the river or transported and released downstream from Bonneville Dam. To this point in the analyses, no significant differences have been observed between return rates for in-river and transported kelts (range of return rates is 6-7%).

Table 11. Project kelt passage (PE), guidance (GE), sluiceway (SLE) and spillway (SPE) efficiencies, sluiceway (SLF) and spillway effectiveness (SPF), at John Day (JDD), The Dalles (TDA), and Bonneville (BON) Dams 2001-2002 (Wertheimer et al. 2003).

Project	Year	N	Spill (%)	PE (%)	GE (%)	SLE (%)	SLF	SPE (%)	SPF
JDD	2002	58	0:54	90	50	NA	NA	79	1.5:1
JDD	2002	97	30:30	95	58	NA	NA	88	2.9:1
JDD	2002	209	NA	93	46	A	NA	87	NA
TDA	2001	28	0	64	NA	64	16.8:1	NA	NA
TDA	2001	70	30	99	NA	89	19.3:1	87	2.9:1
TDA	2002	207	37	95	NA	52	20.0:1	90	2.4:1
BON	2001	73	2	58	53 (B2)	87 (B1)	124.3:1	NA	NA
BON	2001	68	37	88	55	100	349.9:1	54	1.6:1
BON	2002	207	45	90	58	100	250.0:1	67	1.5:1

In spring 2003, Boggs and Peery (2003) examined 1,838 steelhead (using ultrasound) collected from the Lower Granite Dam separator and reported 93% were kelts of which 17% were males and 83% females, and about 50% were wild. They also reported that return rates for kelts PIT tagged and released in 2002 were 2.7% for kelts transported from Lower Granite Dam to the estuary, and 0.8% for kelts released in the Lower Granite Dam tailrace to migrate in-river.

In 2003, mean migration rates of radio-tagged kelts were slower in Snake River reaches (32.4 km/d) than in lower Columbia River reaches (55.2 km/d) (Boggs and Peery 2003). Migration rates were positively correlated with river flow and generally increased in progressively downstream reaches. Of the 112 kelts released in the Lower Granite Dam tailrace, 55% were detected in the tailrace of Ice Harbor Dam and 34% were detected in the tailrace of Bonneville Dam. There was no difference in the migration rates or hydrosystem survival for hatchery or wild kelts.

KEY UNCERTAINTIES ASSOCIATED WITH ADULT PASSAGE

The use of individually coded radiotelemetry tags for adult fish has greatly increased the precision associated with studies of migration behavior at dams and survival through the mainstem corridor. Individual fish have been uniquely tagged, their approach behaviors and passage over dams and through reservoirs monitored, and their run histories reconstructed. A large amount of information has been gathered, and the data from these studies are reported in this document. However, a number of uncertainties associated with adult passage at dams and through the hydropower system remain.

Lower river passage and passage efficiency at Bonneville Dam need to be more fully investigated. Flow control has altered the estuarine hydrograph and may be resulting in delay and/or increased mortality in the estuary. In addition, only fish captured in Bonneville Dam fishways are used to assess passage efficiency at this, the most downstream obstacle to passage. Studies are needed to confirm that passage efficiency estimates at Bonneville are not biased due to the exclusive use of "successful" fish (i.e., those that make it into the fishways where they are captured and tagged). This information is particularly critical if passage recorded at Bonneville Dam PIT-tag detector systems is to be used to compare relative smolt-to-adult survival rates (SARs).

Adult Passage through Turbines

Little information is available regarding the effects on adult Pacific salmon from passing through turbines. The issue has not been discussed in the region or a focus of research until recently. For example, the Corps and Battelle Pacific Northwest National Laboratory recently conducted a turbine passage survival workshop, which included a panel discussion of adult passage (Carlson 2001). Effects on adults associated with passage through turbines can affect upstream migrants from any species that "fallback" through turbines because they overshot their natal tributary or from behavior associated with searching for that stream. It can also affect steelhead kelts that have spawned and are migrating downstream to return to the Pacific Ocean. The topic has two components: what percentage of adults fallback through turbines, and what are the injury and mortality rates associated with this route of passage?

Fallback

Although the rate of adult fallback at dams has been documented, the proportion that can be attributed to dam operations versus migratory behavior has not been determined. A high proportion of fallback behavior is associated with the Bonneville Dam First Powerhouse Bradford Island exit during spill. However, a substantial number of these fish fall back from several miles upstream, and this behavior appears unrelated to dam operations.

Mobile tracking of radio-tagged fish has been conducted in the Bonneville First Powerhouse forebay. This information will be modeled in conjunction with physical models of hydraulic conditions and dam operations to further understand the causes of fallback. Fallback over Ice Harbor Dam of fish destined to the upper Columbia River appears more related to searching for the appropriate tributary than dam operations. Similarly, fallback of fall chinook in the Snake River appears related to fish over-shooting the entrance to Lyons Ferry Hatchery.

Despite the lack of definitive information on the cause of a fall back event, fallback of adults past dams is a serious issue because the mortality associated with this behavior can be high, especially during periods of no spill. Moreover, increased energetic costs associated with migration delays and re-ascending dams may impact reproductive success. For example, Brown et al. (2002) showed a 62% increase in energetic demand for spring Chinook salmon in the tailrace of Bonneville Dam when compared to the forebay (26% increase when compared to the fishway).

Injury and Mortality

Bell (1967) used mathematical formulas to estimate injury and mortality associated with turbine passage. In his formulas, the probability of strike increases with fish length. Thus we assume that mortality rates for adults passing through turbines will be higher than for juveniles. Indeed, based on what data we have, this appears to be the case. Wagner and Foster (1973) evaluated mortality to adult steelhead kelts passing through Kaplan turbines at Foster Dam on the South Santiam River, Oregon. Based on capturing kelts in nets that covered the turbine discharge, they estimated that 22 and 41% of the steelhead kelts they captured died from apparent turbine mortality in 1969 and 1970, respectively.

Normandeau Associates Inc. (2003) evaluated the feasibility of using balloon tags to estimate turbine passage survival of adult salmonids. At McNary Dam in 2002. The evaluated steelhead (432 to 787 mm total length) and rainbow trout (381 to 584 mm total length) and found that 79.2% of the steelhead and 87.5% of the trout were recaptured in the tailrace. They could account for or ascribe a post-passage status to 95.7 and 93.8% of the steelhead and trout, respectively, and concluded it was feasible to use the tags to estimate survival with a fairly high level of precision.

Losses above Lower Granite Dam

Survival of spring and summer chinook salmon from Ice Harbor Dam to spawning grounds or hatcheries varied between years, and was estimated to be no more than 77% and as low as 54% from 1993 and 1991, respectively. Further studies will be required to resolve the accuracy and cause of these preliminary observations.

Reproductive Success

The potential effects migration through the hydropower system has on adult reproductive success is unknown. Successful reproduction requires a migration of both sexes to the spawning grounds, appropriate lipid reserves to carry out the necessary reproductive behavior (nest building and defense, and spawning), high gamete quality, proper embryonic development, and survival of offspring for the downstream migration. Delays, excessive energy consumption, and exposure to higher water temperatures during the migration are factors that could lead to reduced reproductive function, disease, and low quality and quantity of gametes. Studies that address these topics will have to be developed to address this question.

Lamprey and Sturgeon Passage

The potential impacts adult fish passage systems have on non-salmonid species is an uncertainty associated with salmonid passage. Concurrent with the decline of salmonid populations, Columbia River Pacific lamprey (*Lampetra tridentata*) populations have also declined. As an anadromous species, lamprey must pass through the same hydropower system as salmonids. However, they have different migration behaviors and lack the physical swimming capabilities of salmon. Passage success studies show that although approximately 90% of the lamprey released downstream from Bonneville Dam from 1997 to 2000 reapproached the dam, less than 50% passed over successfully and these fish required a median passage time of 4-6 d to do so (Moser et al. 2002). Radiotelemetry studies also indicated that lamprey had the least success in areas with confusing flows, primarily where gratings exist on the floor of the ladder system, and at the serpentine weirs near the tops of ladders. Ongoing studies are needed to address whether the hydropower system is impacting lamprey recruitment, homing, and life histories.

Inter-Dam Losses

Radiotelemetry researchers in the past tagged adult fish from the entire population crossing a given dam, without knowing the source, origin, or evolutionary significant unit (ESU) of the fish tagged. This is a factor when using radio-tagged fish to determine inter-dam losses, since the origin of the fish lost between release and upstream sites was unknown. Recently, the number of PIT-tagged adults returning to the Columbia River is increasing. This, along with the development of adult PIT-tag detection systems at key locations, will enable more precise estimates of survival and inter-dam losses for known-source fish.

Adult Count Accuracy

Adult counting occurs at all mainstem dams to ensure, among other purposes, that fish passage facilities are operating properly. Partial hourly counts are expanded, and little counting occurs during hours of darkness. Counts include all adults passing each dam, and are upwardly biased by any fish that fall back and reascend. The present counting schedule and systems meet their intended purpose of ensuring that the adult passage facilities are functioning properly. However, the ISAB (Bisson et al. 1999) noted a number of problems associated with the precision and accuracy of adult counts at mainstem dams, especially if the data are used to make fisheries management decisions.

Deschutes River Straying

Straying of steelhead into the Deschutes River, OR has been observed during recent radiotelemetry studies. The natal origin of these fish was unknown. The behavior was observed during periods when the water of the Columbia River was warmer than the Deschutes River. Possible causes for this behavior include fish seeking the cooler water, straying behavior associated with transportation, and an evolutionary adaptation that enhances survival. Additional studies of known-source fish will be required to better understand the cause of this observed behavior.

Interspecies Interactions

American shad (*Alosa sapidissima*) are not indigenous to the Columbia River Basin, but this species has successfully exploited the reservoir habitat currently available in the hydropower system. A total of 1.6 million shad passed Bonneville Dam in 1999. The fishery to exploit the shad insurgence resulted in an increased passage time for Chinook salmon as well as an increased rate of fallback (Jepson et al. 2003). Potential biological interactions between shad and salmonids exist but have not been studied. In particular, adult salmonid behavior in fishways may be altered when peaks in shad passage occur during the spring and physical crowding occurs due to high shad densities. Examinations of fine-scale salmonid behaviors in and around the ladders during these periods are needed to assess potential interspecific interactions.

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APPENDIX: NOAA Fisheries Supplemental FCRPS Biological Opinion Spill Program

Appendix Table A1. Estimated spill caps for the operations specified in this supplemental FCRPS Biological Opinion.

Project	Estimated spill level [a]	Hours	Limiting factor
Lower Granite	45 kcfs	6 pm to 6 am	gas cap
Little Goose	60 kcfs	6 pm to 6 am	gas cap
Lower Monumental	40 kcfs	6 pm to 6 am	gas cap
Ice Harbor	75 kcfs (night) 45 kcfs (day)	24 hours	nighttime-gas cap daytime-adult passage
McNary	150 kcfs	6 pm to 6 am	gas cap
John Day	180 kcfs/60%[b]	1 h before sunset to 1 h after sunrise	gas cap/percentage
The Dalles	64%	24 hours	tailrace flow pattern and survival concerns
Bonneville	120 kcfs (night) 75 kcfs (day)	24 hours	nighttime-gas cap daytime-adult fallback

a The estimates of fish passage efficiency (FPE) used to derive these spill levels are conservative and based on fish guidance efficiencies (FGE) of hatchery spring/summer chinook salmon, rather than wild or hatchery steelhead, to protect the weakest listed stock present during the steelhead outmigration period.

b The total dissolved gas cap at John Day Dam is estimated at 180 kcfs and the spill cap for tailrace hydraulics is 60%. At project flows up to 300 kcfs, spill discharges will be 60% of instantaneous project flow. Above 300 kcfs project flow, spill discharges will be 180 kcfs (up to the hydraulic capacity of the powerhouse).